



Understanding and reducing the impact of fishing on discarded sharks
and rays with innovative new tools for conservation and research

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Abstract

A key aim of applied marine conservation research is to understand and manage the impact of human activities on marine biodiversity. Sharks and rays (elasmobranchs) form an important component of marine systems and they provide important ecosystem services. However, declines in elasmobranch populations are being documented in fisheries around the world. We are still developing an understanding of the full impacts of fishing on elasmobranchs and many of the effects of fisheries discards on their populations are still unresolved. Little consideration has been given to the potentially pervasive effects of capture-induced parturition (premature birth or abortion) on elasmobranch populations. Capture-induced parturition is an issue faced by elasmobranchs caught as bycatch in fisheries targeting other species. Bycatch and discarding is of concern for a number of fishing methods, but this thesis focuses on gill-net fisheries. More specifically those gill-nets deployed extensively in a number of countries as a strategy to cull sharks for the protection and safety of bathers. Managers have few options to reduce this bycatch and there are few socially accepted or proven alternative methods to mitigate shark incidents. I aimed to address these knowledge gaps by using a variety of methods and by pioneering new techniques.

First, using a systematic review of the literature I define, examine and present analyses on the prevalence of capture-induced parturition. I sought to determine the species and species traits that make elasmobranchs most susceptible to this phenomenon. I found capture-induced parturition to be a prevalent but under-recognised issue for live-bearing elasmobranchs. It occurs commonly across an assortment of species, particularly live bearers, in response to a wide variety of fishing methods. Those species impacted include a number that are threatened or endangered. Marine protected areas (MPAs) are one of the few management tools available to address this issue and also offer a unique means of testing for fishing impacts. I explored the movement patterns of a commonly discarded elasmobranch *Trygonorrhina fasciata* to assess habitat requirements and space use in relation to marine protected areas. I found that *T. fasciata* utilises relatively small areas ($< 1.12 \text{ km}^2$) for extended periods, indicating that current no-take areas seem adequate for this species. I then explored the threat of capture-induced parturition by comparing the abundance, size and pregnancy rates of *T. fasciata* in fished and no-take areas. I found elevated pregnancy rates and higher abundance of juveniles in no-take areas compared to fished areas,

suggesting that capture-induced parturition in fished areas is having impacts on recruitment. These higher pregnancy rates and apparent increase in recruitment also suggest that marine protected areas may offer a tool to buffer these capture-induced impacts for those elasmobranchs that move on spatial scales relevant to area-based protection.

Second, I introduce aerostats (blimps) as an ecological sampling technique which provide continuous monitoring of the nearshore environment with the potential to offer a non-invasive and socially acceptable alternative to gill-nets as a shark-mitigation strategy. I found this new aerial surveillance method to be an effective and socially preferred alternative to the high bycatch associated with shark meshing, offering a new tool for human safety, shark conservation and ecological research.

My thesis represents an important step in improving our understanding of the impacts of fishing by quantifying the prevalence and effect of capture-induced parturition on discarded elasmobranchs. It also confirms the utility and importance of MPAs in shark conservation. In a global context, this work represents an important contribution to the conservation and responsible management of elasmobranchs.

This thesis is dedicated to the late Dr Neil Adams.

Thank you for the gift of curiosity.

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List of Publications included as part of this thesis


This thesis has been prepared in journal article compilation style format. Chapter 2 has been published and has been recognised as a being among the most influential papers published in the Journal Biological Conservation (see *Pakeman et al., 2019).

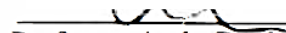
Paper	Chapter	Status	Journal
Adams, K. R., Fetterplace, L. C., Davis, A. R., Taylor, M. D., Knott, N. A., (2018). Sharks, rays and abortion: the prevalence of capture-induced parturition in elasmobranchs. <i>Biological Conservation</i> , 217, p. 11-27, https://doi.org/10.1016/j.biocon.2017.10.010	2	Published	Biological Conservation
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Adams, K. R., Fetterplace, L. C., Wosnick, N., Davis, A. R., Taylor, M. D., Knott, N. A. (in prep) Pregnant and Protected: no-take marine reserves can buffer the impact of capture-induced abortion in discarded elasmobranchs	4	In prep.	Proceedings of the National Academy of Science
Adams, K. R., Broad., A., Ruiz-Garcia, D., Davis, A. R. (2020)) Continuous wildlife monitoring using blimps as an aerial platform: a case study observing marine megafauna. <i>Australian Zoologist</i> : 2020, Vol. 40, No. 3, pp. 407-415. https://doi.org/10.7882/AZ.2020.004	5	Published	Australian Zoologist
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*Pakeman, R.J., Bates, A., Corlett, R.T., Cumming, G.S., Johns, D., Koh, L.P. , Loyola, R. , Maas, B., Pejchar, L., Primack, R.B. Regan, T.J., Roth, R., Godet, L., Descoteaux, D., Devictor V., 2019. Fifty years of biological conservation *Biol. Conserv.*, 230, pp. A1-A4,

Statement of Candidate Contribution

As the primary supervisor, I, Professor Andy Davis, declare that the greater part of the work in each article is attributed to the candidate, Kye Rhys Adams. In each of the papers that constitute this thesis, Kye led conceptual development, study design and was primarily responsible for the data collection, data analysis and data interpretation. The first draft of each manuscript was written by the candidate who was then responsible for responding to the editing suggestions of his coauthors. The co-authors were responsible for assisting with the study design, data collection, interpreting data and editing manuscripts where necessary.


Kye Rhys Adams
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Date: 19/6/19


Professor Andy Davis
Principal Supervisor
Date: 19/6/2019

Additional Publications and Presentations

In addition to the manuscripts listed above, during the course of my PhD I have co-authored three journal articles and presented data from my thesis at eight national and international conferences.

Published papers

Swadling, D. S., Knott, N. A., Rees, M. J., Pederson, H., Adams, K. R., Taylor, M. D., & Davis, A. R. (2020). Seagrass canopies and the performance of acoustic telemetry: implications for the interpretation of fish movements. *Animal Biotelemetry*, 8(1), 1-12.

Wosnick, N., Awruch, C.A., Adams, K.R., Gutierre, S.M.M., Bornatowski, H., Prado, A.C., Freire, C.A. (2018) Impacts of fisheries on elasmobranch reproduction: high rates of abortion and subsequent maternal mortality in the shortnose guitarfish, *Animal Conservation*

Wosnick, N., Adams, K.R., Freire, C.A. (2018) Ultrasonography as a promising methodology to indicate captured-induced abortion in viviparous elasmobranchs, *Journal of fish biology* 93 (6), 1033-1037

Conference presentations

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Adams, K. R., Knott, N. A., Gibbs, L., Hing, M., Davis, A. R. (2017) Project AIRSHIP: Spotting sharks and rays using blimp- mounted cameras for conservation and human safety, *The 10th Indo-Pacific Fish Conference*, 2-6 October, Tahiti.

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Certification

I, Kye Rhys Adams, declare that this thesis, submitted in partial fulfilment of the requirements of the award of Doctor of Philosophy, in the School of Biological Sciences, University of Wollongong, is wholly my own work unless referenced or acknowledged. The document has not been submitted for qualifications at any other academic institution.


Kye Rhys Adams
PhD Candidate
Date: 19/6/19

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Appendix 6 – links to science communication material related to this thesis

Chapter 1 – General Introduction

1.1 The impact of fishing on sharks and rays

Declines in many elasmobranch populations have been documented in fisheries around the world, attributable to both targeted elasmobranch fisheries and fisheries where elasmobranchs are caught as bycatch (Stevens, 2000; Dulvy et al., 2008; Dulvy et al., 2014). The impact of fishing on many sharks and rays is exacerbated by life-history characteristics which put them at an immediate and elevated risk from overfishing. Their low fecundity, late maturation and slow growth rates reduce their recovery potential from disturbances such as fishing, particularly if it overlaps with a large proportion of the species range and is prolonged (García et al., 2008; Dulvy et al., 2008; Field et al., 2009; Dulvy et al., 2014). The impact of fishing on elasmobranchs changes among species due to variation in life history traits and differential exposure to fishing activity. Large, late maturing species with high spatial overlap with fisheries appear to be most vulnerable (Stevens et al., 2000; Dulvy et al., 2014; Queiroz et al., 2019). Most of the chondrichthyan species identified as threatened are found in depths of less than 200 m, which is most likely due to a combination of high intensity fishing pressure coupled with habitat alteration/destruction in nearshore areas (Dulvy et al., 2014). Approximately one quarter of sharks, skates and rays are considered threatened by International Union for Conservation of Nature (IUCN) standards (Dulvy et al., 2014) which is largely due to targeted and incidental catch. There are ongoing challenges with monitoring the catch of sharks and rays. If recorded at all, catches are often misidentified, aggregated, or discarded, and in many parts of the world unregulated, which presents difficulties for monitoring and management (Clarke et al., 2006; Bornatowski et al., 2013; Dulvy et al., 2014).

Although global catch records remain patchy, there is increasing evidence that fishing has direct effects on the abundance, size/age structure and population dynamics of elasmobranchs (Stevens, 2000; Dulvy and Reynolds, 2002; Dulvy et al., 2008). The most compelling evidence for direct fishing impacts comes from the declines of target elasmobranch populations as a result of both commercial and recreational fishing (Heupel et al., 2004; Bensley et al., 2010). Due to the animal being landed, mortality rates are easily measurable for target species, or those taken as byproduct (species caught as bycatch which are landed for sale) and such harvest can be biologically sustainable (Simpfendorfer and Dulvy, 2017). In comparison, it is harder to estimate and manage the impacts of fishing on discarded species due to the difficulties in measuring post-capture mortality and other effects that manifest post-release (Molina and Cooke, 2012).

There is considerable evidence that discarding impacts species that interact with fishing gear and can result in sub-lethal effects such as stress and injury or lead to direct mortality (Rodrigues et al., 2018). Some of the declines of the most threatened elasmobranch species, such as sawfishes, have been due to incidental capture (Dulvy et al., 2014).

1.2 The prevalence of incidental catch and discarding for sharks and rays

Incidental capture and discarding is a major concern for many elasmobranchs with high rates of capture in both commercial and recreational fisheries. A common problem shared by almost all fishing gear, from industrial trawlers to the hand line, is the inherent lack of selectivity. Regrettably, there is collateral damage to non-target species captured incidentally and this potential impact needs to be quantified and managed. In commercial fisheries, the historical recording of species-specific discards is notoriously poor (Stevens et al., 2000; FAO, 2019). Shark discards, however, have undergone some assessment, with Worm et al. (2013) estimating a global discard amount totalling 1,135,000 tonnes of sharks for the year 2000 with survival dependant on post-capture mortality and finning rates. Discarding of sharks and rays can be even higher among recreational fishers with ‘catch and release’ fishing being fairly common practice. For example, sharks are among the top 10 key recreationally fished species in the US with > 95% being discarded (FEUS, 2015). In Australia, elasmobranchs species are commonly caught and often discarded by recreational fishers; an estimated 81% of the 1,252,728 sharks and rays caught in the year 2000 were discarded (Henry and Lyle, 2003; Dulvy et al., 2014).

Species with low resilience to fishing can be severely affected by being caught as bycatch in commercial and recreational fisheries (Molina and Cooke, 2012; Dulvy et al., 2014). The management of bycatch species is inherently more complex than that of targeted species as much of the discarded catch is not documented; many discarded species have unknown mortality rates, and incidentally caught species can shift between being sold as by-product and being discarded (Molina and Cooke 2012). Discarding can have a number of potential impacts ranging from post-capture mortality (PCM) to reduced fitness caused by stress and injury (Wilson et al., 2014; Guida et al., 2017; Rodrigues et al., 2018). PCM is difficult to estimate because it depends on the fishing method used and on the species interacting with the fishing gear (Musyl et al., 2011; DAFF, 2013; Dapp et al., 2015). This unknown post capture mortality (PCM) is of concern given the magnitude of the catch, and the documented vulnerability of the species involved. Reduced fitness caused by the stress of capture is even more complex to test for, however there is new evidence that capture-stress can reduce the birthweight of embryos (Guida et al., 2017) and can even lead to the loss of offspring through stress-induced abortion (Wosnick et al., 2018a; Wosnick et al.,

2018b). Whether these responses to being fished and discarded are leading to population level impacts and contributing to elasmobranch declines remains unresolved. Quantifying and understanding the full impact of fishing on elasmobranch species is a requirement for designing adequate management plans for elasmobranchs (Worm et al., 2013). One means of testing for fishing impacts, and potentially managing impacted species is marine protected areas, which are increasingly being implemented for conserving biodiversity worldwide (Gaines et al., 2010).

1.3 Marine protected areas as a conservation tool and study system

Marine protected areas (MPAs) are spatial closures to fishing that provide a valuable study system for determining and potentially managing anthropogenic impacts such as fishing. There continues to be rapid increases in the amount of area under protection globally (Ward-Paige and Worm, 2017). MPAs are primarily targeted at conserving biodiversity, and restrictions to fishing are one way of doing this. They also typically have other restrictions in place which minimise pollution and prevent habitat loss by restricting development as well as reducing anchoring impacts (Gaines et al., 2010; MPMAP, 2015). In combination, these strategies are designed to provide the habitat and assemblage structure necessary to maintain biodiversity and possibly provide a source of recruitment to surrounding areas via ‘spill over’ (Roberts et al., 2001; Gaines et al., 2010). Although Marine Protected Areas often show positive effects on species (Roberts et al., 2001), the magnitude and extent of most effects depend largely on the rate and scale of animal movement in relation to reserve size (Kramer and Chapman, 1999). A spatial closure to fishing, such as a no-take marine reserve, is unlikely to prove effective if the movement of species cover areas much larger than the area closed to fishing (Fetterplace et al., 2016). Highly mobile or migratory species may move in and out of these areas frequently and as a result MPAs, or individual no-take reserves within them, may be too small to protect a species that may only spend small portions of time ‘protected’. Consequently, to ensure the design of MPAs is adequate to protect elasmobranchs, a detailed understanding of their movement patterns and spatial requirements is necessary (Chapman et al., 2005; Block et al., 2011; Papastamatiou and Lowe, 2012).

The movement behaviour of elasmobranch species has been shown to be highly variable on both spatial and temporal scales (Heupel et al., 2004; Brodie et al., 2018). Many elasmobranch species are highly migratory, with some covering thousands of kilometres within a year (Heupel et al., 2004). However, there is evidence of philopatric behaviour in various species (Speed et al., 2010; Chapman et al., 2015), particularly in species with defined nursery areas (Heupel et al., 2007). Due to philopatry, and the high investment of females into a small number of offspring, identification of such nursery areas is fundamental to the effective management of

elasmobranch species (Kelaher et al., 2015). Once identified, strategies such as the restriction of fishing may be employed to reduce the mortality of females during gestation and early life-history stages (Otway et al., 2004; Mourier and Planes 2013; Kelaher et al., 2015). The level of protection provided by MPAs is likely to be greater for elasmobranchs that have smaller home ranges, and which show site association for longer periods (Speed et al., 2016). There is a growing body of research supporting hypotheses that no-take marine reserves can help conserve certain shark populations, particularly those belong to the family *Carcharhinidae* (Bond et al., 2012). These studies can be divided into those which show differences in relative abundance between reserves and fished sites (e.g. Meekan and Cappel 2004; Heupel et al., 2009) and those which demonstrate that reef sharks reside inside reserve boundaries (e.g. Hueter et al., 2005; Garla et al., 2006; Heupel et al., 2010; Escalle et al., 2015; Acuña-Marrero et al., 2017; Chapman et al., 2015) but very few studies do both (Bond et al., 2012). Tagging studies of *Manta birostris* (Dewar et al., 2008) *Dipturus batis*, (Wearmouth and Sims, 2009) and *Raja clavata*, (Hunter et al., 2006; Walker et al., 1997) indicate high levels of site attachment suggesting that some batoids may also benefit from the protection of no-take marine reserves (Wearmouth and Sims, 2009). It remains undetermined whether the MPAs are effective for species that are discarded and determining if reserve size is appropriate for at risk bycatch species requires targeted attention.

In addition to conserving biodiversity, MPAs provide an opportunity for an independent assessment of the effects of recreational and commercial fishing. Where no-take marine reserves have been randomly or haphazardly positioned, such areas can be compared to fished areas provide an assessment of the effects of fishing activity. Experiments across MPA zones and boundaries borrow heavily from manipulative experiments with the aim of making more reliable inferences about the causes of observed changes in fish populations (Fetterplace, 2011). Baum and Worm (2009) argue that such experiments are therefore more experimental than observational. In such experiments, continued fishing becomes the ‘treatment’ against which control areas can be compared in order to ascertain any impacts of fishing. Differences in the size and abundance of species when comparing representative fished and unfished areas would indicate that fishing was impacting these species and allows for the species-specific magnitude of effect to be determined. Alternatively, a lack of difference between these areas may indicate that fishing is having minimal impact on species or the no-take reserve is too small to protect species. A spatial and temporal comparison of fished areas and no-take areas can provide an assessment of the impacts of fishing and contribute to the sustainable management of ecosystems (Gaines et al., 2010). There has been extensive research into understanding and quantifying the effect of MPAs on the general biotic assemblages within their boundaries (Gaines et al., 2010).

As a result, there is an increasing body of scientific evidence demonstrating that the abundance, diversity and length of targeted fish species increase inside MPAs when compared to areas that continue to be fished (Willis et al., 2003; Alcala et al., 2005, Kelaher et al., 2014). What remains unclear is whether the abundance, diversity and length of discarded elasmobranchs increase inside no-take marine reserves which would reveal the full impact of fishing on these species.

1.4 The need for alternatives to high bycatch shark-control programs

Despite their threatened status, shark populations are often deliberately controlled due to their perceived risk to people, fishing gear, and other fisheries (Dulvy et al., 2014). This practice has contributed to the threatened status of at least 12 species (Dulvy et al., 2014). Long-established shark-control programmes exist for bather safety off New South Wales (since 1937) and Queensland (1962), Australia, and KwaZulu-Natal, South Africa (1952) (Stevens et al., 2000). The key aim of these shark control programs is to reduce the numbers of potentially dangerous large sharks from coastal waters. They do this by using fishing gear to catch and kill sharks. Two fishing methods are common practice; gill-nets and baited drumlines (Green et al., 2009). I focus on these nets for my current research as they have the highest rates of bycatch. These gill-nets do not enclose beaches but are typically 150 metres in length and set several hundred metres from shore. When first deployed in a new location, catch rates in these nets generally show a rapid decline, and then become stable, with some variation between species and locations (Stevens et al., 2000). They are known to have high levels of bycatch and in Australia the program is listed as a Key Threatening Process under both the Fisheries Management Act 1994 and Threatened Species Conservation Act 1995 in recognition of its impacts on threatened species and other non-target animals. Over 70% of animals caught in these nets are non-dangerous bycatch species (DPI, 2009) and over the past 60 years close to 400 Critically Endangered Grey Nurse sharks have been killed by the NSW shark meshing program (DPI, 2009).

Bycatch is a well-known issue for gill-nets which are a common fishing strategy used commercially and artisianally as well as for shark management. During normal use, gill-nets are known to capture and kill large numbers of non-dangerous sharks and rays, seabirds, sea mammals, and sea turtles (Waugh et al., 2011; Wiedenfeld et al., 2015). In some cases gill-net bycatch is suspected to drive population declines (Wiedenfeld et al., 2015). Minimization of non-target catch is a key component of fishing responsibly (Wiedenfeld et al., 2015). Mitigation methods have been proven effective for other fishing gears, but methods to reduce gill-net bycatch have had much less development (Wiedenfeld et al., 2015). Many mitigation techniques rely on species traits (e.g. pingers for deterring echolocating species; Mackay and Knuckey, 2013) and

while they may work for one species or species group, they may prove ineffective for another (Waugh et al., 2011). Due to gill-nets operating by entanglement of animals and due to this inherent lack of selectivity, it is unlikely that bycatch can be completely eliminated in nets targeting sharks for population control. It is difficult to develop strategies that selectively avoid catching non-dangerous sharks given their similarities in size and behaviour to ‘dangerous’ species. Perhaps the only mitigation method in such a case is to find an alternative strategy for bathers safety that does not involve controlling shark populations by deploying gill-nets.

Given shark control programs are centred on preventing sharks interacting with humans, the human element is important to quantify. Social acceptance is key to the success of any approach with peoples feeling of safety a key consideration. In high-tourism coastal regions, culling policies for managing human-shark encounters are increasingly prevalent, although these measures lack social acceptance or definitive evidence for their effectiveness (Green et al., 2009; Cliff and Dudley, 2011; McPhee, 2012; Gibbs and Warren, 2014; Gibbs and Warren, 2015). Gibbs and Warren (2015) show that even after a number of shark incidents in Western Australia the majority of ocean-users opposed the kill-based shark hazard management strategies. The ocean-users surveyed strongly supported further research focusing on understanding shark behaviour and approaches that enable people to understand and accept risks associated with ocean use. Denyer-Simmons and Mehmet (2018) also show growing support for detection, deterrent and surveillance technologies. Much like using marine protected areas to reduce human impacts such as fishing, methods that eliminate the reliance on destructive strategies for shark management are preferable. Innovative solutions are required to minimise environmental impact and maximise risk reduction. An ideal solution would provide positive coexistence and conservation outcomes, ensuring sustainability, and quantified levels of risk and social acceptance. Emerging technologies offer the potential to provide such a solution and render the need for shark meshing obsolete.

1.5 Thesis Aims and Structure

There continues to be high amounts of bycatch and subsequent discarding in elasmobranchs. In contrast to targeted fisheries, the full impact of fishing on species that are commonly discarded remains poorly studied. Marine protected areas provide a perfect study system by providing reference locations with varying levels of fishing and offer a potential management strategy to mitigate fishing impacts. There is a clear need for research focusing specifically on the effects of MPAs on the size, abundance and reproduction of discarded elasmobranchs. In addition, acquiring data on movement and behaviour is essential to effective MPA design and management. In this thesis, I explore the impacts of fishing on discarded sharks and rays and develop ways to

mitigate bycatch. I also propose a solution which disrupts the reliance on gill-netting for shark management; eliminating the need for a high bycatch fishing method. This thesis has two main aims: (1) to assess the impact of fishing, focusing on capture-induced abortion, on discarded elasmobranchs and (2) to develop a novel and non-lethal alternative to the predominant and lethal shark meshing strategy currently employed for bather safety; thereby offering a means to reduce bycatch.

In **chapter 2** I explore the prevalence of capture-induced parturition in elasmobranchs. I systematically review and present analyses on this largely unreported, often misinterpreted and poorly understood impact of fishing on these animals. I aim to synthesize a diffuse body of literature to estimate the prevalence and rate of capture-induced parturition. I aim to investigate the influence of species, threat-levels, method of capture, reproductive mode and gestation extent on the occurrence of capture induced parturition. In a novel approach I also aim to use social media to identify videos depicting capture-induced parturition events to provide supplementary visual evidence for the phenomenon. The mortality of embryos could have implications for elasmobranch populations, and there are limited options to deal with this problem.

In **chapter 3** I describe the movement patterns of *Trygonorrhina fasciata* to determine how they utilise habitat within and beyond protected areas. Currently, the space use of rays (batoids) is poorly understood and management of batoids requires targeted study. This is particularly relevant in soft-sediment habitats, as it is often assumed that species in this homogenous environment have little reason to show residency or site attachment and are therefore wide-ranging. With the success of marine protected areas dependant largely on the rate and scale of animal movements I aim to determine the long-term movement behaviour of this batoid species and how it varies with life-history and abiotic factors.

Chapter 4 explores the impact of capture-induced abortions on recruitment, and consequently population size and structure. Due to high maternal investment and long gestation periods this source of mortality may mean that the potential impacts are substantial at a population level. No-take marine reserves provide an opportunity to assess the impact of this inter-generational mortality by offering reference locations where fishing is prohibited. Assessing movements is a precursor for interpreting species response to protection. I aim to determine long-term residency of a commonly discarded elasmobranch (*Trygonorrhina fasciata*) to a no-take area. I also aim to determine whether capture-induced parturition affects recruitment by comparing the size, abundance and pregnancy rates in areas with and without fishing.

Chapter 5 introduces my proposal to use of blimps as an aerial platform for continuous wildlife monitoring. Aerial surveys are a powerful means of collecting ecological data in terrestrial and marine systems that may otherwise be difficult to acquire. Increasingly aerial observations are made with Unmanned Aerial Vehicles (UAVs), such as drones. As this technology has improved in reliability and affordability it has replaced the traditional use of fixed-wing aircraft and helicopters. Drones have limitations; primarily in their limited flight endurance, their potential to disturb wildlife and concerns over safety. Here I introduce an aerostat (a ground tethered blimp) as a logistically simple and economical alternative to drones and other aircraft. Blimps differ from drones by using helium for lift, thereby conserving battery life. I aim to determine the utility of my novel technique to enhance ecological research. To do this I use my safe and accessible alternative aerial platform to observe the foraging behaviour of wildlife including sharks and seals for extended periods; demonstrating its utility for a range of applications including potentially improving human safety.

In **chapter 6** I expand on the potential of using my novel aerial surveillance method to provide a non-lethal alternative to traditional shark mitigation methods. I aim to assess aerial surveillance using a blimp as a technique to simply and effectively reduce shark encounters at ocean beaches and determine the social acceptance of this technique as compared to an established hazard mitigation strategy – shark meshing. If shark spotting rates are high with this technique, it could provide a measurable reduction in risk from sharks, improving beach safety and facilitating coexistence between people and wildlife. Conflict between humans and sharks is a longstanding challenge that can present negative consequences for humans and marine ecosystems. Developing non-lethal strategies for mitigating the risk sharks to humans facilitates sustainable ecosystem management by reducing bycatch and the need to control shark populations.

1.6 Study location and model species

To assess the impact of fishing on discarded elasmobranchs I conducted my research in New South Wales (NSW), Australia. The NSW Marine Parks Authority plans and manages a system of marine parks (MPA, 2009). Within each park is a network of no-take sanctuary zones which exist within a larger system of zones where restricted fishing is allowed. In NSW, sanctuary zones are the equivalent of no-take marine reserves and Marine Parks are the equivalent of MPAs in other regions. Jervis Bay Marine Park was established in 1998 with the management plan commencing in October 2002. There are four management zones across the park (MPA, 2009). Sanctuary zones make up approximately 20% of the park and prohibit all forms of fishing and collecting. Habitat protection zones make up 72% of the marine park with recreational fishing and some very limited forms of commercial fishing permitted. General use zones cover 8% of the

marine park and are where commercial and recreational fishing are allowed, provided that they are ecologically sustainable. Special purpose zones account for 0.2% and are generally marinas or mooring areas. The Marine Park Authority bought out many of the commercial fishing licences, so recreational fishing is now the main fishery in the Bay (Lynch, 2006). Notably, the Independent Scientific Audit of Marine Parks in New South Wales (Beeton et al., 2012) recommends that recreational fishing (including distribution of effort, catch, discards of bycatch and ecosystem impacts) must be evaluated and results need to be incorporated into future marine park management plans. To develop and test the novel surveillance method (blimps) I conducted the work in the town of Kiama, NSW, Australia with location details outlined in the relevant chapters (see chapter 5 section 2 and chapter 6 section 2 for details).

To determine the effects of incidental catch and discarding, species of the order Rhinobatiformes provide a good model due to high discard rates in Australia, particularly species in the groups Rhinobatidae and Rhinidae (shovelnose rays and guitar fish). A 2010 recreational fishing survey (Taylor et al., 2010) estimated up to 97% discard rates for these species groups ($22,067 \pm 6,588$ SE caught and $21,305 \pm 6,534$ SE discarded). Within these groups, the most abundant species in my study area is the eastern fiddler ray *Trygonorrhina fasciata* (Fetterplace, 2011). This species is a ubiquitous species endemic to eastern Australia between southern QLD and Twofold Bay, NSW. It has been observed in Jervis Bay on reefs, in the shallow subtidal, in seagrass and over soft sediment areas (CSIRO 1994; Broad et al., 2010; Kiggins 2013; Rees, 2017). In commercial fisheries, this ray was ranked 16/172 species of fishes caught by trawling in a study of the benthic fish fauna in temperate waters off the west coast of Australia (Hyndes et al., 1999). Despite having broad geographic range and being relatively common, Baker (2011) identifies the species group as potentially vulnerable to decline. The closely related southern fiddler ray *Trygonorrhina dumerilii* is reported to have a low resilience to exploitation based on reproductive parameters (Baker, 2011), and high vulnerability to fishing-induced population impacts (Cheung et al., 2005). The benthic habit and apparent preference for nearshore, shallow-water habitats make the species among those most vulnerable to capture by a variety of fishing methods (Dulvy et al., 2014; Baker, 2011). Fiddler rays have relatively low fecundity compared to other rhinobatids with an average of 3 embryos produced per breeding cycle (Marshall et al., 2007). The eastern fiddler ray *T. fasciata* has size at birth of less than 250 mm (Last and Stevens 2009) and a maximum reported total length of 1200 mm (Huveneers, 2015) with females growing to greater lengths than males (Marshall et al., 2007). Recent ecological risk assessments have ranked the species group as being at high risk of population level impacts in the South East Trawl fishery and Great Australian Bight Trawl fishery, due to capture and discarding in large numbers (Baker,

2011). There remains little fishery independent information about *Trygonorrhina* spp. population sizes over space and time. There is also uncertainty in regards to NSW fisheries dependent data, due to aggregation of bycatch into species groups in records. Therefore, the numbers caught by recreational and commercial fishers remain uncertain. There are some location-specific estimates of numbers caught (e.g. Hyndes et al., 1999), but there have been no assessments of the significance of these catches over any spatial or temporal scale in fished and non-fished areas (Baker, 2011).

1.7 References

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Chapter 2 – Sharks, rays and abortion: the prevalence of capture-induced parturition in elasmobranchs

2.0 Abstract

The direct impacts of fishing on chondrichthyans (sharks, rays and chimeras) are well established. Here we review a largely unreported, often misinterpreted and poorly understood indirect impact of fishing on these animals — capture-induced parturition (either premature birth or abortion). Although direct mortality of discarded sharks and rays has been estimated, the prevalence of abortion/premature birth and subsequent generational mortality remains largely unstudied. We synthesize a diffuse body of literature to reveal that a conservative estimate of > 12% of live bearing elasmobranchs (n = 88 species) show capture-induced parturition. For those species with adequate data, we estimate capture-induced parturition events ranging from 2 to 85% of pregnant females (average 24%). To date, capture-induced parturition has only been observed in live-bearing species. We compile data on threat-levels, method of capture, reproductive mode and gestation extent of premature/aborted embryos. We also utilize social media to identify 41 social-media links depicting a capture-induced parturition event which provide supplementary visual evidence for the phenomenon. The mortality of embryos will have implications for elasmobranch populations, and there are limited options to deal with this problem. This review is the first to synthesize available data on capture-induced parturition in sharks and rays, and highlights an important ethical and management issue for fishers and managers deserving of much greater attention.

2.1 Introduction

Sharks, rays and their relatives (chondrichthyans; Table 2.1) are some of the slowest growing and oldest maturing vertebrate animals (Dulvy et al., 2014). They also exhibit some of the highest levels of maternal investment and longest gestation periods in the animal kingdom (Cortés, 2000; Dulvy et al., 2014). These combined life-history traits make them sensitive to overfishing and many population declines have been observed (e.g. Graham et al., 2001; Stobutzki et al., 2002; Cortés et al., 2007; Oliver et al., 2015). The direct effects of both targeted and incidental capture of chondrichthyans has been the focus of much directed research including numerous reviews (Stevens et al., 2000; Frisk et al., 2005; Worm et al., 2013; Dulvy et al., 2014; Oliver et al., 2015). Although direct fishing mortality is of concern, capture-induced abortion/parturition (Table 2.1) is a less obvious, but potentially pervasive effect on the reproductive potential of many sharks and rays. We define capture-induced parturition as any birthing event prompted by interaction with fishing gear. A capture-induced parturition event is either a premature birth or an abortion depending on the gestation extent of embryos (Table 2.1; Fig. 2.1).

Table 2.1 - List of terms and definitions as referred to in this review.

List of definitions	
Stress-induced parturition	The premature birthing of offspring stimulated by physiological processes which involve a response to a 'stressor' via complex pathways modulated by hormones.
Capture- induced parturition	The birthing of offspring prompted by interaction with fishing gear. The pathways stimulating birth are expected to vary and could be caused by interactions between physical trauma and physiological stress.
Premature birth	The parturition of pre-term offspring, which often have reduced fitness due to lack of development and smaller body size.
Abortion	The termination of a pregnancy by the expulsion of a fetus or embryo before it can survive outside the uterus.
Shark	Shark generally refers to those elasmobranchs with gill slits located laterally and includes all the nine orders of chondrichthyans that are not deemed skates, rays or chimeras: sawsharks (Pristiophoriformes), angel sharks (Squatiniiformes), dogfish (Squaliformes), sixgilled sharks (Hexanchiformes), mackerel sharks (Lamniformes), ground sharks (Carcharhiniiformes), carpetsharks (Orectolobiformes), bullhead sharks (Heterodontiformes) and bramble sharks (Echinorhiniiformes).
Ray	Ray generally refers to the three orders of chondrichthyans that are not deemed sharks, skates or chimeras. All rays are live-bearing and have gill slits that are located ventrally: stingrays (Myliobatiformes), electric rays (Torpediniiformes) and shovelnose rays/guitarfish (Rhinopristiiformes).
Skate	Skate refers to all species in the order Rajiformes. All skates are egg-laying and have gill slits that are located ventrally.
Chimera	Chimera refers to all species in the order Chimaeriformes, a cartilaginous fish order that together with the elasmobranch orders makes up the chondrichthyan class of fishes. All chimeras are egg-laying and have gill slits that are located laterally.

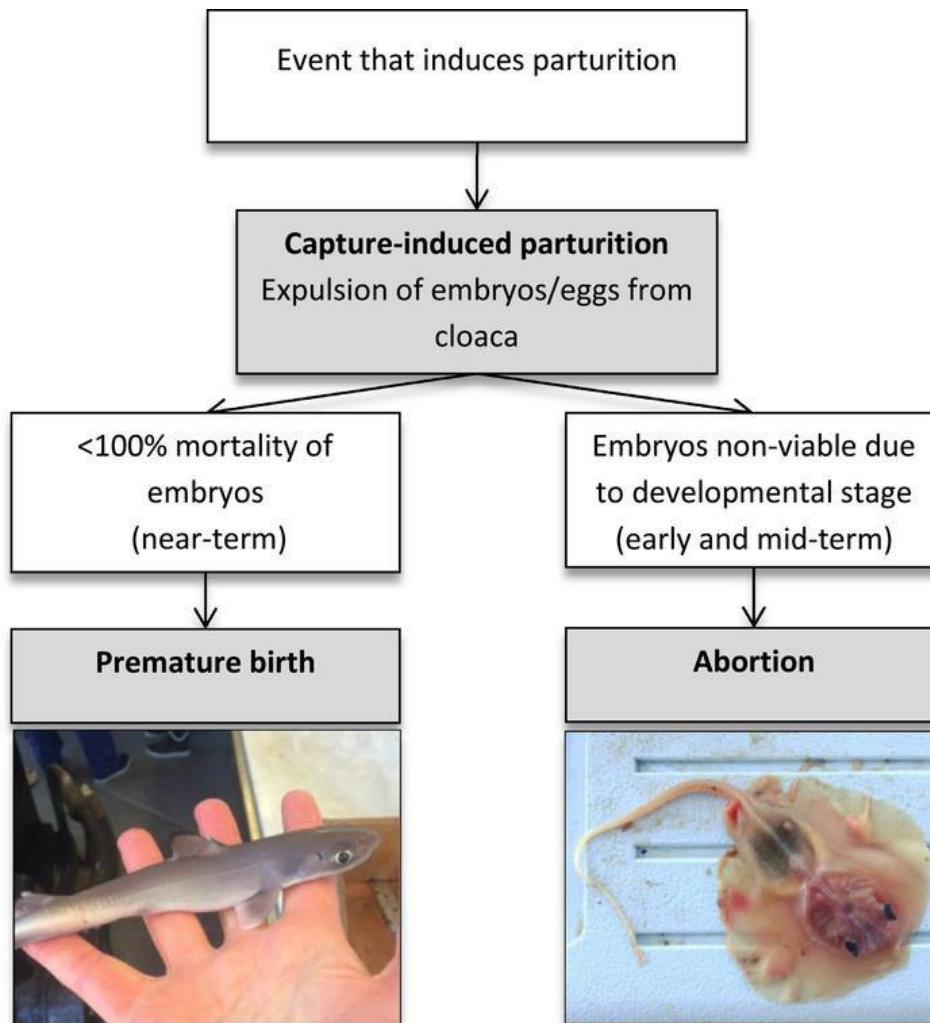


Figure 2.1- Decision tree showing the terminology relating to capture-induced parturition used in this review. Left image shows a near-term spikey dogfish (*Squalus megalops*) of approximately 24 cm total length. Photographer: L. Fetterplace. Right image shows an early term Atlantic stingray (*Hypanus sabinus*) embryo that is notably pale. The yolk sac is not present but may have ruptured during parturition. Photographer: C. Collatos.

2.1.1 Defining capture-induced parturition

Capture-induced parturition in sharks and rays is by no means a novel phenomenon; there are numerous anecdotal observations in the scientific literature, some of which date to over 200 years ago. The phenomenon has so far attracted very little interest, other than sporadic references to the inconvenience it causes when measuring fecundity (e.g. Ebert, 1984; Struthsaker, 1969; Snelson et al., 1988). There seems to be a general lack of awareness among recreational fishers of the occurrence of capture-induced parturition in sharks and rays (see Table A1.2). There is also a distinct lack of targeted research into the occurrence and cause of capture-induced parturition, making it difficult for managers to incorporate into bycatch management. Our suspicion is that these casual reports, when viewed as a whole, indicate that capture-induced parturition is a common event with potential impacts on the reproductive capabilities of species. This may lead to effects on recruitment in shark and ray populations.

The earliest record of capture-induced parturition we have identified was by Risso (1810):

*“A female of *Squatina*, of a considerable size, taken from our net, gave fifteen to twenty pups at the time where, due to lack of water, it was asphixed by the action of the atmosphere on its gills.”*

It is therefore surprising that 200 years later the phenomenon of capture-induced parturition remains unstudied and unquantified in any detail, other than sporadic observations and reports. Although it has been noted that fecundity in elasmobranchs is sometimes difficult to estimate because they abort their young on capture (Struthsaker, 1969), we are yet to develop a clear understanding of the frequency, specific cause, and impact of these “abortions”. We know of no studies that have been specifically interested in capture-induced parturition beyond incidentally observing and recording it other than Wosnick et al. (2018a) and Wosnick et al. (2018b).

The phenomenon of capture-induced parturition in elasmobranchs has been noted in the literature under a variety of terms, including ‘abortion’, ‘capture-induced abortion’, ‘spontaneous abortion’, ‘slip’, ‘sudden parturition’, ‘dropping young’ and ‘premature birth’. Given that nothing is currently known about the survival of embryos after the event, ‘abortion’ may not correctly describe the process in all cases, given that some near-term offspring may survive. We propose that “capture-induced parturition” is the most suitable blanket term for the process, with capture-induced abortion most appropriate for cases where complete embryo mortality occurs (Fig. 2.1; Table 2.1). Importantly, until mortality estimates for these embryos are determined, application of the precautionary principle (Lauck et al., 1998) suggests that all capture-induced parturition events should be viewed as capture-induced abortions (i.e. all pups are assumed to die). We also propose that ‘spontaneous abortion’ is not an appropriate characterisation given that it ignores the fact that the parturition events are capture-induced and may not be spontaneous.

To recognize a capture-induced parturition, premature pups (see Figs. 48 and 54 in Babel, 1966 for example photographs of the different gestation stages of *Urolophus halleri*) are visibly protruding from the cloaca or present on deck (Fig. 2.2; Table A1.1). It is important to differentiate capture-induced embryos from captured young-of-the-year given that many fishing methods may mix embryos with small juveniles. For *U. halleri*, with a three-month gestation period, the yolk sac is almost fully absorbed approximately two weeks before birth (Babel, 1966), which offers a simple method to determine gestation extent. This time frame for yolk sac absorption may differ for species with longer gestation periods. An important consideration is that chondrichthyan embryos tend to acquire most species characteristics by the middle of the gestation period (Babel, 1966, Fig. 48), which could lead to them being mistakenly reported as full-term (Pratt and Casey, 1990) especially if there is no known size-at-birth for the species. Upon dissection, a distended uterus with broken uterine compartments can also

indicate that a capture- induced parturition has occurred (Pratt and Casey, 1990), however, this method cannot exclude the possibility of a recent natural birth.



Figure 2.2- Capture-induced parturition event in the spikey dogfish (*Squalus megalops*) caught via demersal trawl off eastern Tasmania, Australia. The head of the near-term pup can be seen protruding from the cloaca (top). The bottom photograph shows the same pup alongside the mother. Photographer: L. Fetterplace.

Stress appears to be a key contributing factor that induces parturition/abortion given that such births have been reported to occur following various methods of fishing, stranding (Williams et al., 2010) and possibly an unsuccessful predation event (Marshall and Bennett, 2010). Parturition has also been observed after administration of anesthetic (Ferreira, 2013; Silbernagel and Yochem, 2016), injection of quinine (Rall and Zubrod, 1962), during an inter-uterine endoscopy (Carrier et al., 2003) and during a sonogram (Mollet et al., 2002). It remains unclear, however, whether it was these specific procedures or the stress on the animal that induced these parturitions. In fishery capture-induced parturition, common stress-inducing stimuli include physical trauma (e.g. harpoons, netting injuries) or asphyxiation (e.g. caught in mesh net, left on deck). The physical trauma and physiological stress caused by capture is likely to vary with fishing method and the sensitivity of the species involved (Dapp et al., 2015). The nature and magnitude of stress responses are species-specific, and linked to physiology as

well as the form and intensity of the stressor (Skomal and Mandelman, 2012). We know that fishing can cause major stress to sharks and their relatives, however the species-specific thresholds that induce parturition remain undetermined.

Stress-induced parturition events do not appear to be isolated to capture. The fact that they can occur in nature means that the phenomenon may have adaptive significance. The earliest record of abortion in sharks and rays appears in the fossilized embryos of a Devonian chondrichthyan (*Delphyodontos dacrifformes*), with a yolk sac still attached but lacking an adult nearby, dated 318 m.y.a. (Lund, 1980). There is further evidence of abortion occurring in the now extinct *Harpagofututor volsellorhinus* ~ 318 m.y.a. (Grogan and Lund, 2011). Stress-inducing stimuli that exist in nature may include stranding, predation attempts, toxic dinoflagellate blooms, thermal shock and hypoxia.

2.1.2 Sensitivity of chondrichthyans to fishing

Sharks, rays and their relatives share a number of life-history traits which make them particularly sensitive to overfishing. Bycatch, or incidental capture is a major concern for many of the approximately 1145 species of elasmobranchs (sharks, rays and skates) and 49 species of holocephalans (chimeras), which together comprise the chondrichthyes class of fishes (Oliver et al., 2015; Naylor and Davies, 2017). For species caught as bycatch that are commonly discarded, the impact of fishing is often assumed to be low if post-capture mortality is low (Oliver et al., 2015). This may not always be the case, with capture-induced parturition representing a potential source of generational mortality for discarded species. The low fecundity and low natural mortality of many sharks and rays leads to a close relationship between the number of pups produced and the size of the breeding population (Stevens et al., 2000). Due to a combination of slow growth rates and late maturation, overfished shark populations can take decades to recover (Stevens et al., 2000).

Even for well managed fisheries with monitored bycatch levels, we lack a definitive list of which species give birth on capture, and the frequency at which it occurs. In fisheries where the majority of shark, ray and chimera species are discarded (e.g. Braccini et al., 2012; Henry and Lyle, 2003) the process of capture-induced parturition has the potential to lead to mortality of recruits, even if the mature female survives the capture event. Although the individual survival of many discarded species may be high (Braccini et al., 2012), pregnant females that give birth during or after capture will lose some, if not all, of their pups for that reproductive cycle. With some elasmobranch species having gestation periods of 2 years or more (e.g. *Squalus acanthias*, Ford, 1921), an abortion event represents a major loss of maternal investment. Pregnant females of some species are known to aggregate seasonally in shallow, warm waters, (e.g. *Triakis semifasciata*, Nosal et al., 2013), making them especially vulnerable to fisheries capture.

Capture-induced parturition is only expected to be problematic and potentially unmanaged when pregnant females are discarded alive rather than landed. Recording of species-specific ray and skate discards is notoriously poor on a global scale (Stevens et al., 2000). Shark discards, however, have undergone some assessment, with Worm et al. (2013) estimating a global discard amount totaling 1,135,000 t of sharks for the year 2000. Further, Worm et al. (2013) estimate that 80% of these discarded sharks were finned and subsequently died, while the remaining 227,000 t were released alive. Of these live discards, many that were pregnant had the potential to lose pups. If current and proposed management strategies for shark finning are implemented, such as banning finning at sea, the number of sharks discarded alive would likely increase, and so too would the potential for capture-induced parturition to impact these species. Recreational fishing also causes capture-induced parturition, and has the potential to affect shark and ray populations given the often high levels of catch and release for these species. For example, an estimated 81% of the 1,252,728 sharks and rays caught over a 12 month period by Australian recreational fishers were discarded (Henry and Lyle, 2003).

2.1.3 Reproductive modes and maternal provisioning in chondrichthyans

The different reproductive modes in chondrichthyans may influence the potential and impact of capture-induced parturitions. Sharks and their relatives can be subdivided into two main reproductive groups; live-bearing (viviparous ~ 700 species) and egg-laying (oviparous ~ 500) (Fig. 2.3). Although all sharks develop inside an egg case, the eggs of live-bearing species ‘hatch’ inside the uterus while egg-laying species hatch externally. An important distinction is that all live-bearers fall within Elasmobranchii. Elasmobranchs have also been categorized into two further modes of reproduction based on maternal provisioning. These two means of nutrient delivery are matrotrophy, where embryonic development is supported by additional maternal input of nutrients; and, lecithotrophy, where development is sustained wholly by a yolk-sac (Wourms, 1981) (Fig. 2.3).

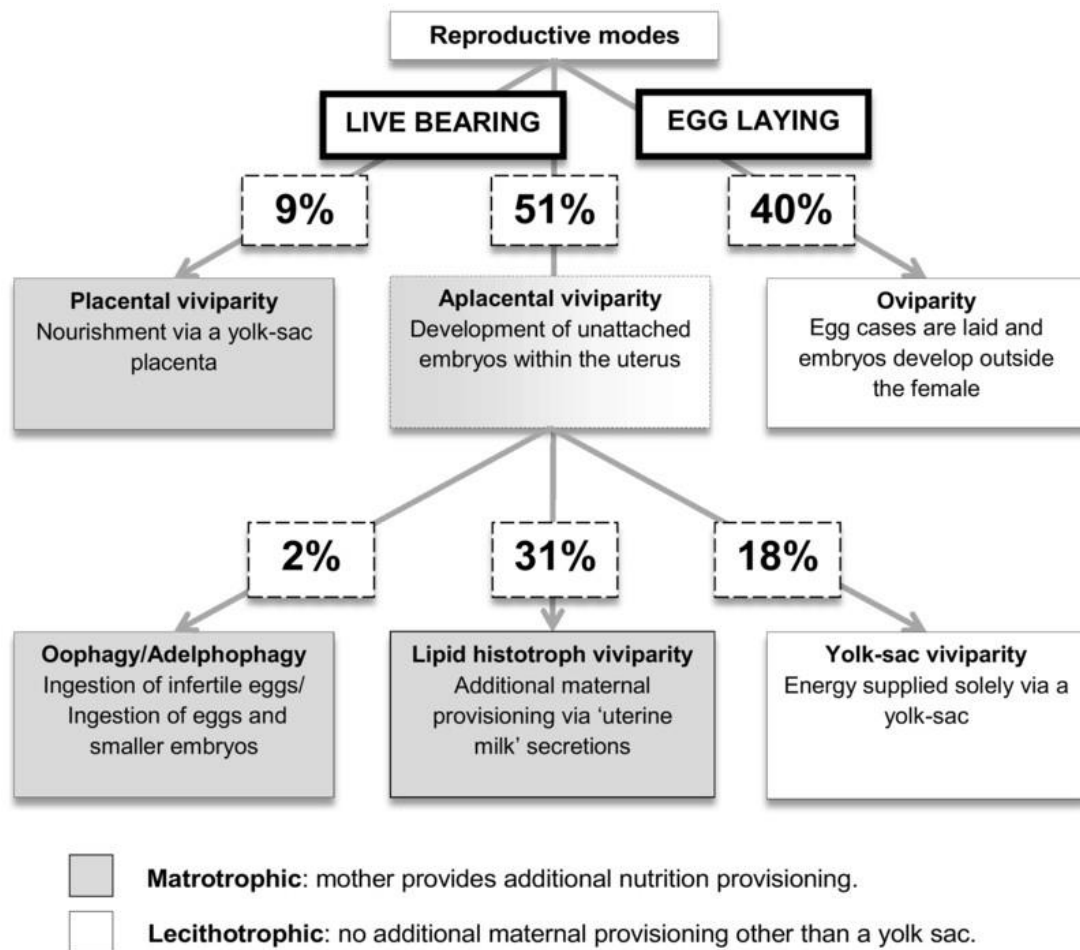


Figure 2.3 - Patterns of reproduction in chondrichthyan species with subdivisions by reproductive modes and maternal input. Percentages in each reproductive category are taken from Dulvy and Reynolds (1997).

To improve the general understanding of capture-induced parturition/abortion in chondrichthyan species, we present the first systematic review of the prevalence of the phenomenon in the reproductive literature on sharks and rays. To understand how prevalent capture-induced parturition is across chondrichthyan species, we have compiled a list of species that are known (or are inferred) to have experienced parturition once they have been captured (Adams, 2017). To help direct future research and conservation efforts, we assess whether some species groups have been reported to experience capture-induced parturition more or less frequently than expected by chance. From a subset of the papers identified in our literature search, we estimate the rate of capture-induced parturition for a number of elasmobranch species caught using a range of fishing methods. This rate represents an estimate of the percentage of pregnant females of a species that give birth on capture. This rate also provides the first assessment of the potential impact of capture-induced parturition on commonly discarded species. Additionally, we investigate whether reproductive mode affects the occurrence of capture-induced parturition. To assess whether the occurrence of abortion may correlate with increased extinction risk we test whether capture-induced parturition may correlate with higher IUCN threat levels. We briefly

explore different fishing methods and how stress and subsequent pup mortality during capture may be reduced. The analysis incorporates data from a wide range of species and locations and therefore provides a worldwide synthesis of capture-induced parturition in chondrichthyans. We also use reports from social media to further assess parturition across sharks and rays in relation to recreational and commercial fishing. Videos which depict suspected induced parturition events provide anecdotal and supplementary evidence to scientists, and provide a novel source to document capture-induced parturition which is independent of the scientific literature and represents ‘real world’ occurrences of these events. Furthermore, we supplement this dataset with our own images documenting capture-induced parturition events during our own scientific investigations and compile a list of other such videos found on social media. Finally, we highlight areas for further research and provide recommendations for researchers and fishers to reduce the chance of causing stress and inducing parturition.

2.2 Methods

2.2.1 Literature search

In order to compile a list of species that exhibit capture-induced parturition/abortion we used structured literature searches. These searches were conducted using Google Scholar, Scopus and Web of Science in June 2017. The Boolean (AND/OR) search terms used in Google Scholar consisted of:

Chondrichthyes AND abort and elasmobranch AND abort

These search terms were not applicable in Scopus and Web of Science as these two databases can only locate search terms in titles and abstracts, unlike Google Scholar, which searches whole texts. Given the lack of targeted literature it was rare for ‘abortion’ to be mentioned in either the title or abstract. We deemed an alternative search strategy was therefore necessary for Scopus and Web of Science.

The modified Boolean (AND/OR) search terms used in subsequent Web of Science and Scopus searches consisted of the following:

Stingray* OR ray OR shark OR skate OR wedgefish OR guitarfish OR batoid* OR elasmobranch* OR chondrichthy* AND reproduc* OR fecundity

After nuisance terms were removed via term filters and duplicates were accounted for, these searches identified 314 texts in Google Scholar, 168 texts in Web of Science, and 168 texts in Scopus. The results of these searches were examined for any references to the abortion of embryos. Any relevant references cited in these papers that were not identified in the database searches were also included. All relevant references were examined for the species, capture method, gestation stage of the embryos, and reproductive mode.

Species were categorized into those in which capture-induced parturition was directly observed (n = 139 instances) or those in which we inferred capture-induced parturition (n = 92 instances). These observed and inferred capture-induced parturitions included multiple reports of individual species. Observations of capture-induced parturition were categorized by the presence of eggs or embryos either in nets, on the deck of fishing vessels or seen being expelled from pregnant females. Inferences of capture-induced parturition were usually based on comments from the author, noting empty and distended uteri after capture, or reference to abortion in related species.

The threat level of each species known to experience capture-induced parturition was determined using the search function in the IUCN Redlist of Threatened Species (IUCN, 2016). If species were not listed on the IUCN Redlist they were assigned a separate category of Not Evaluated. The IUCN Redlist currently lists 1095 species, however Naylor and Davies (2017) lists 1194 species. This review uses the Chondrichthyan Tree of Life (Naylor and Davies, 2017) for species classification and numbers of shark and ray species.

2.2.2 Calculating estimates of the frequency of capture-induced parturition

To estimate the rate of capture-induced parturition, one of two methods was used depending on the data available. Twenty-six studies included adequate information to estimate capture-induced parturition rates. Each estimate provides a rate of parturition for a single species within a study for the reported fishing method. The criteria for the inclusion of a study required reporting of data that satisfied both the numerator and denominator of either of the following two equations:

- 1) The number of females reported to abort compared to the total number of gravid females in the study:

$$\frac{\text{Number of females that showed induced parturition}}{\text{Total number of gravid females (pregnant + 'induced')}}$$

Fourteen observations of a single gravid female showing capture-induced parturition were excluded (see Adams, 2017) as these estimates would inflate the abortion frequency (i.e. 100%).

- 2) The proportion of embryos resulting from capture-induced parturition (usually on deck) was compared to the total number of embryos reported in the study (in uteri embryos were determined via dissection):

$$\frac{\text{Number of embryos from induced parturition (on deck)}}{\text{Total number of embryos (in uteri + on deck)}}$$

If either the numerator or denominator of either formula could not be satisfied then the study could not be included in the estimate of the capture-induced parturition frequency. All four categories were not reported for any study, presumably because once embryos are on deck it is impossible to identify which

female they came from. This means the number of females that showed induced parturition cannot be known once embryos are on deck or free in the net. Both methods described would underestimate the true capture-induced parturition rate due to unobserved capture-induced parturition events leading to a loss of embryos before landing or after release. Eggs resulting from capture-induced parturition were not included in our estimates.

2.2.3 Categorizing capture-induced parturition events by reproductive mode, embryo's developmental stage and method of capture

To determine if reproductive mode may influence the occurrence of capture-induced parturition, reports were classified into the following categories (Fig. 2.3):

1. placental viviparous
2. oviparous
3. aplacental viviparous (yolk-sac)
4. aplacental viviparous (histotroph)
5. aplacental viviparous (oophagy/adelphophagy)

Reproductive mode was determined from the literature using either the original literature search reference, the IUCN threat assessment (IUCN, 2016), or from Compagno (1990). To gain an understanding of the range of gestation extents of embryos, the developmental stage was noted as described in the paper in which capture-induced parturition was reported. Importantly, although eggs were aborted, these came from live-birthing (viviparous) species and were presumed to be very early stage or unfertilized.

The fishing method/s used in each study were classified into 12 categories based on the studies in which capture-induced parturition was observed. These 12 categories were artisanal fishing, gill-nets, harpoon, hook- and-line, longlines, net (unspecified), seine-net, gunshot, tangle net, trawling, multiple and unspecified. Some studies reported outcomes from multiple fishing methods; therefore, the method being used when capture-induced parturition was observed was unable to be determined.

2.2.4 Compiling anecdotal observations of capture-induced parturition

In order to supplement parturition events documented in the literature, videos of capture-induced parturition events were compiled via Youtube, Instagram and Facebook. We used variations of the search terms “ray birth” and “shark birth” and also the related video algorithms provided by these networks. Only videos with live females actively aborting were included. The number of views, location and suspected species were also recorded. Confident species identification was not possible in many

cases due to the quality of the video, the lack of adequate viewing angles and limited geographical information.

2.2.5 Statistical analyses

To identify species groups with higher than expected observations of parturition, and thus those species/taxa that might be particularly vulnerable, we calculated the number of species expected in each group if we assume capture-induced parturition is equally likely to be reported across all species groups. In order to generate an expected value for a given category (e.g. order, family, IUCN category, reproductive mode) it was assumed that all taxa had an equal probability of being drawn. If a subset of shark or ray species were drawn randomly from all chondrichthyans (in our case a subset of 88), the total number of species in each category can be used to predict the number expected in the smaller subset. For example, the Order Myliobatiformes contains 217 of the 1194 extant chondrichthyan species i.e. 18% (Naylor and Davies, in prep). Using this ratio we expect 16 from a random draw of 88 chondrichthyans will be Myliobatiformes i.e. our expected value. These expected values were compared to the observed number of species showing capture-induced parturition in each category by using exact tests of goodness-of-fit with a Monte Carlo approach ($n_{\text{trial}} = 1e + 7$, $atOnce = 1e + 6$). For calculating expected frequencies, we assumed that all species were equally likely to experience capture-induced parturition due to fishing. This requires the assumption that all species within taxa were equally exposed to fishing. Those species groups identified by such analyses provide a clear starting point for targeted research; however the cause of such patterns is open to interpretation and could be the result of innate vulnerability or sampling bias. All chondrichthyan species (species = 1194) were used to calculate the expected distribution of capture-induced parturition based on the number of species in each order (from the Chondrichthyan Tree of Life). For the family analysis, only families with at least one species showing capture-induced parturition were used (species = 535). To determine whether those species which show capture-induced parturition experience the same threat levels as other elasmobranchs, we also used an exact test of goodness-of-fit using a Monte Carlo approach ($n_{\text{trial}} = 1e + 7$, $atOnce = 1e + 6$) with the expected distribution calculated from the IUCN red list (species = 851 see Table A1.5). Finally, we used an exact test to determine whether capture-induced parturition was more or less frequent in each of the four live-bearing modes of reproduction.

Those categories driving any differences indicated by the exact tests were determined using post hoc tests. Analyses were performed in R 2.14.2 (R Core Team, 2016) using the EMT package (Menzel, 2013) `multinomial.test()` function for Goodness-of-Fit Test for Discrete Multivariate data using methods from Mangiafico (2016). Post hoc tests were conducted using the `binom.test()` function using methods specified by Mangiafico (2015). Chi square goodness-of-fit tests were not used due to the number of expected values below 5.

2.3 Results and discussion

2.3.1 The prevalence of capture-induced parturition in the scientific literature

Our literature search collected 139 reports of 88 species from 26 families directly observed to exhibit capture-induced parturition (see Adams, 2017 for the full species list). Capture-induced parturition was only observed in live-birthing (viviparous) species and, to date, does not appear to have been reported in the scientific literature for egg-laying (oviparous) species. We note that 12% ($n = 88$ species) of live-bearing elasmobranch species have been observed to show capture-induced parturition. If the additional species which have been inferred in the literature to exhibit capture-induced parturition are confirmed, the prevalence of capture-induced parturition could increase to 18% ($n = 127$ species) of live-bearing elasmobranchs.

Capture-induced parturition was reported more frequently than expected in the Orders: Myliobatiformes (stingrays; observed: 39/88 expected: 16/88, $p < 0.001$), Lamniformes (mackerel sharks; observed: 4/88 expected: 1/88, $p = 0.018$) and Squatiniformes (angel sharks; observed: 8/88 expected: 1/88, $p < 0.001$) (Fig. 2.4a). Capture-induced parturition was reported less frequently than expected for the Orders: Rajiformes (skates; observed: 0/88 expected: 22/88, $p < 0.001$) and Chimaeriformes (chimeras; observed: 0/88 expected: 4/88, $p = 0.035$) presumably because all species in these orders are egg-laying.

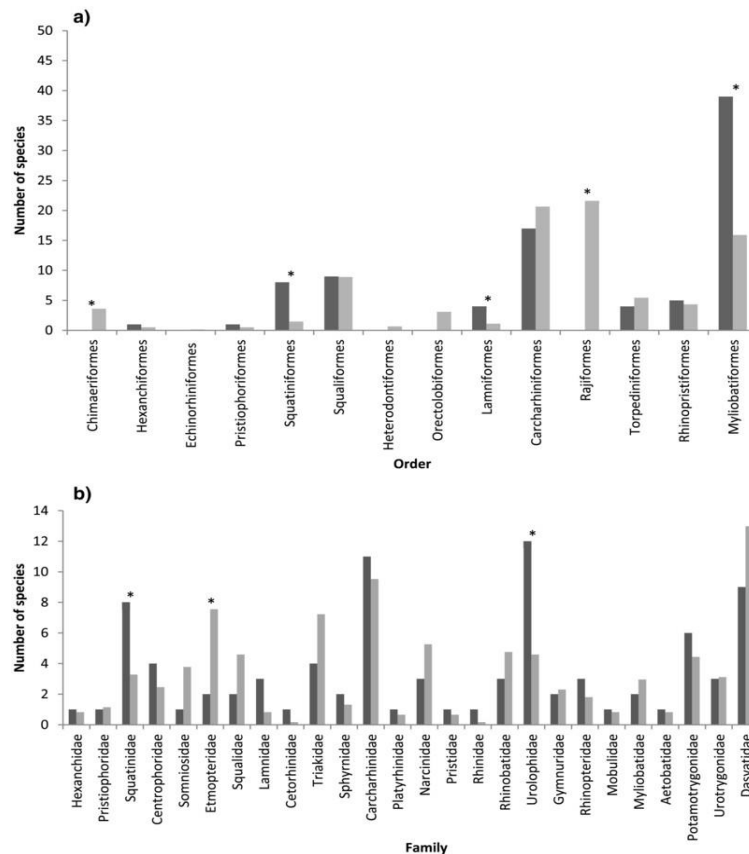


Figure 2.4- a) Order level classification of chondrichthyans comparing the number of species observed to experience (black bars) and expected to experience capture-induced parturition if it occurs independent of order classification (grey bars). Expected frequencies are calculated using the number of species in each order (from Naylor and Davies, 2017) compared to the total number of chondrichthyans ($n = 1194$). Significant differences ($\alpha < 0.05$) are denoted by *.

b) Family level classification of elasmobranchs for the 26 families which contain species observed to experience capture-induced parturition. The number of species observed (black bars) is compared to the number of species expected (grey bars) if capture-induced parturition were independent of species classification. Expected frequencies are calculated using the number of species in each family (from Naylor and Davies, 2017) compared to the total number of species in families exhibiting capture-induced parturition ($n = 536$). Significant differences ($\alpha < 0.05$) are denoted by *.

Family level analysis showed capture-induced parturition to be reported more frequently than expected in the Families: Urolophidae (stingarees; observed: 12/88 expected: 5/88, $p = 0.004$), and Squatinidae (angel sharks; observed: 8/88 expected: 3/88, $p = 0.011$) (Fig. 2.4b). Importantly, angel sharks are the second most threatened family of chondrichthyans after sawfishes (Dulvy et al., 2014). Interestingly, in the Family Etmopteridae (lantern sharks) capture-induced parturition was reported less frequent than expected (observed: 1/88 expected: 6/88, $p = 0.024$). We can only speculate that this may be due to the majority species in this deep-water family being poorly-known and rarely encountered (Kyne et al., 2007).

Whether these groups experience capture-induced parturition more commonly due to shared traits or whether the induced parturition occurrence is an artefact of sampling bias requires further analysis. The

IUCN (2016) red list shows only 11% of Urolophidae and 25% of Squatinidae are Data Deficient compared to 51.8 % of all skates and rays (Dulvy et al., 2014). This may indicate that these two families are relatively well studied and the high number of observed species experiencing capture-induced parturition is an artefact of sampling effort for these two families. It is indeed possible that these two families have received greater attention regarding capture-induced parturition. This is indicated by the fact that Osaer et al. (2015) give references to all 8 species of Squatinidae known to experience capture-induced parturition. For Urolophidae, 9 of the 12 reports of capture-induced parturition come from papers where White or Trinnie are either a lead or a co-author. This may indicate that these two species groups have experienced a form of bias whereby one observation has led to an apparent increase in capture-induced parturition being reported. Alternatively, Squatinidae (angel sharks) are known to exhibit cloacal gestation whereby embryos complete their development within a uterine-cloacal chamber, which is open to the exterior via the cloacal vent (Sunyem and Vooren, 1997). This form of gestation may contribute to the apparent increased occurrence of capture-induced parturition in this family (Sunyem and Vooren, 1997) and deserves further research.

Given that parturition was not isolated to capture (see Williams et al., 2010; Marshall and Bennett, 2010) we suggest that stress-induced parturition may have adaptive significance. We propose two hypotheses for the occurrence of this phenomenon in chondrichthyans:

- 1) The self-sacrifice hypothesis; whereby a pregnant female is stranded by wave action or a retreating tide and gives birth to increase the chances of survival of her pups and ensure continuation of her genes. Induced parturition due to stranding has only been documented in the literature for *Hexanchus griseus* (Williams et al., 2010).
- 2) The predation/self-preservation hypothesis; a pregnant female when stressed gives birth to facilitate her escape and potentially the survival of the remainder of her litter. For example, stress-induced parturition due to a predation attempt has been inferred for *Manta alfredi* (Marshall and Bennett, 2010). As a terrestrial comparison, female kangaroos have the tendency to drop their pouch-young if they are stressed by a predator (Ealey, 1963), or during capture and handling (NHMRC, 2014). Low (1978) theorizes that a female who deliberately abandons her offspring is more likely to escape predation due to the loss of encumbrance and the diversion created by the abandoned young.

2.3.2 The frequency of capture-induced parturition events

The average capture-induced parturition frequency across 26 studies, covering 24 species, was 24% (Table 2.2). This indicates that where data are available, ~ 2 in 10 gravid females across a subset of species showed capture-induced parturition. The rate of parturition is, however, quite variable among species (Table 2.2); it ranged from 2% of embryos being induced in *Carcharhinus brevipinna* (Capapé et al., 2003) and *Galeocerdo cuvier* (Jaquemé et al., 2013) to 85% of females releasing embryos on

capture for *Pteroplatytrygon violacea* (Mollet, 2002). Given this variability, the induced parturition rate is likely to be highly species specific; however there are currently insufficient data to support reliable conclusions given the variation in fishing methods used across these studies.

Table 2.2 - The frequency of capture-induced parturition calculated for 26 species from a variety of fishing methods.

Common name	Species	Capture-induced event	Total number of gravid females or embryos	Frequency (%)	Fishing method	Author/s
Pelagic stingray	<i>Pteroplatytrygon violacea</i>	41 capture-induced parturitions	48 gravid females	85	Longlines	Mollet, 2002
Ornate angel shark	<i>Squatina tergocellata</i>	32 capture-induced embryos	50 total embryos	64	Trawling	Bridge et al., 1998
Common stingray	<i>Dasyatis pastinaca</i>	27 capture-induced parturitions	45 gravid females	60	Multiple	Saadaoui et al., 2015
Thorny stingray	<i>Dasyatis centroura</i>	3 capture-induced parturitions	5 gravid females	60	Trawling	Struthsaker, 1969
Scaly whipray	<i>Brevitrygon imbricata</i>	1 capture-induced parturition	2 gravid females	50	Longlines	Henderson and Reeve, 2011
Pelagic stingray	<i>Pteroplatytrygon violacea</i>	2 capture-induced parturitions	5 gravid females	40	Hook-and-line	Siqueira and Sant'Anna, 2007
Banded stingaree	<i>Urolophus cruciatus</i>	51 capture-induced embryos	145 total embryos	35	Multiple	Trinnie, 2013
Venezuela round stingray	<i>Urotrygon venezuelae</i>	10 capture-induced parturitions	35 gravid females	29	Seine-net	Acevedo et al., 2015
Roger's round ray	<i>Urotrygon rogersi</i>	113 capture-induced embryos	582 total embryos	19	Trawling	Mejía-Falla et al., 2012

Common name	Species	Capture-induced event	Total number of gravid females or embryos	Frequency (%)	Fishing method	Author/s
Spotted stingaree	<i>Urolophus paucimaculatus</i>	106 capture-induced	553 total embryos	19	Multiple	Trinnie, 2013
Blue shark	<i>Prionace glauca</i>	6 capture-induced parturitions	37 gravid females	16	Longlines	Tavares et al., 2012
Finetooth shark	<i>Carcharhinus isodon</i>	2 capture-induced parturitions	13 gravid females	15	Multiple	Castro, 1993
Blacktip shark	<i>Carcharhinus limbatus</i>	6 capture-induced embryos	40 total embryos	15	Gill-net	Capapé et al., 2004
Dusky smooth-hound	<i>Mustelus canis</i>	2 capture-induced parturitions	15 gravid females	13	Longlines	Zagaglia et al., 2011
Blue shark	<i>Prionace glauca</i>	2 capture-induced parturitions	15 gravid females	13	Longlines	Hazin et al., 1994
Sawback angelshark	<i>Squatina aculeata</i>	1 capture-induced parturitions	8 gravid females	13	Multiple	Capapé et al., 2005
Sandyback stingaree	<i>Urolophus bucculentus</i>	15 capture-induced embryos	128 total embryos	12	Multiple	Trinnie et al., 2012
Birdbeak dogfish	<i>Deania calcea</i>	2 capture-induced parturitions	18 gravid females	11	Multiple	Irvine et al., 2012
Bluespotted stingray	<i>Neotrygon kuhlii</i>	1 capture-induced embryo	10 total embryos	10	Multiple	Pierce, 2009

Common name	Species	Capture-induced event	Total number of gravid females or embryos	Frequency (%)	Fishing method	Author/s
Brown guitarfish	<i>Rhinobatos schlegelii</i>	1 capture-induced parturition	10 gravid females	10	Multiple	Schluessel et al., 2015
Eastern shovelnose stingaree	<i>Trygonoptera imitata</i>	9 capture-induced embryos	115 total embryos	8	Multiple	Trinnie et al., 2009
Greenback stingaree	<i>Urolophus viridis</i>	6 capture-induced embryos	83 total embryos	7	Multiple	Trinnie et al., 2015
Round stingray	<i>Urobatis halleri</i>	1 capture-induced parturition	22 gravid females	5	Seine-net	Jirik and Lowe, 2012
Sandbar shark	<i>Carcharhinus plumbeus</i>	2 capture-induced embryos	46 total embryos	4	Gill-net	Cliff et al., 1988
Tiger shark	<i>Galeocerdo cuvier</i>	1 capture-induced embryo	43 total embryos	2	Unspecified	Jaquemet et al., 2013
Spinner shark	<i>Carcharhinus brevipinna</i>	2 capture-induced embryos	88 total embryos	2	Multiple	Capapé et al., 2003

An important consideration in estimating the frequency of capture-induced parturition is the time of year when captured. For example, we have observed two species experiencing capture-induced parturition in the same trawl; one species released near-term pups (premature birth) and the other species released mid-term embryos (abortion) (authors', pers. obs.). This is indicative of the importance of reproductive seasonality determining the potential impact of a capture-induced parturition based on the extent of gestation. Future estimates of the frequency of capture-induced parturition should attempt to factor in the reproductive periodicity of different species to provide a temporal estimate that may inform managers when considering temporal closures.

2.3.3 Correlates of capture-induced parturition

2.3.3.1 IUCN threat levels

The occurrence of capture-induced parturition does not appear to correlate with a heightened threat of extinction in elasmobranch species. There are, however, considerably fewer species reported than expected in the Data Deficient (DD) (observed: 17/88 expected: 33/88, $p < 0.001$) category (Fig. 2.5). This is concerning since the majority of chondrichthyans are currently DD ($n = 475$; IUCN, 2016). This means that they have not been studied in great detail, and extrapolation points to a potentially large proportion of the ~ 700 live-bearing species exhibiting capture-induced parturition in response to fishing. During the assessment of these species, we would encourage publication of observations of any species which show capture-induced parturition, and that an estimation of the frequency be included if possible. Capture-induced parturition was observed more frequently (20/88) than expected (10/88) for species in the Near Threatened category ($p = 0.002$, Fig. 2.5). Although induced parturition does not appear to correlate with higher threat levels, there are 7 species known to exhibit capture-induced parturition that are either Critically Endangered or Endangered (Table 2.3).

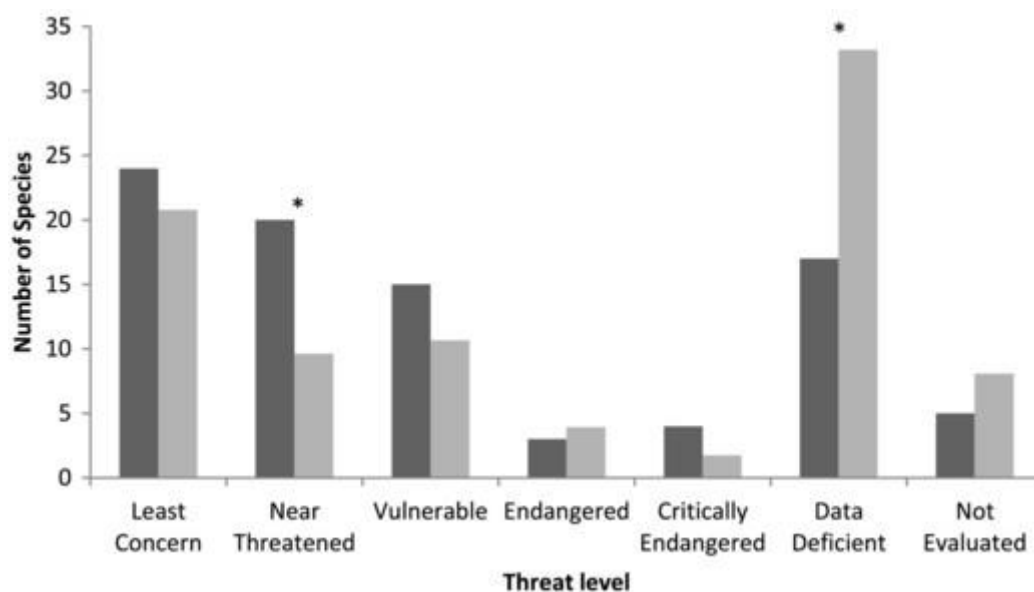


Figure 2.5 - The number of chondrichthyan species in each IUCN category for elasmobranchs that experience capture-induced parturition (black bars, $n = 88$ species) compared to the distribution expected calculated from all 851 elasmobranchs that are not classified as either skates or chimeras (grey bars). See Appendix 1 Table A1.5 for the number of species in each IUCN category. Significant differences are denoted by *.

Table 2.3 - Species listed as endangered or critically endangered on the IUCN red-list that are known to exhibit capture-induced parturition.

Common name	Species	IUCN category	Author/s
Large-tooth sawfish	<i>Pristis pristis</i>	Critically endangered	Marden, 1944
Sawback angel shark	<i>Squatina aculeata</i>	Critically endangered	Capapé et al., 2005
Angel shark	<i>Squatina squatina</i>	Critically endangered	Risso, 1810
Caribbean electric ray	<i>Narcine bancroftii</i>	Critically endangered ^a	Carvalho et al., 2007
Scalloped hammerhead shark	<i>Sphyrna lewini</i>	Endangered	Clarke, 1971
Argentine angel shark	<i>Squatina argentina</i>	Endangered	Cousseau, 1973
Coastal stingaree	<i>Urolophus harrisi</i>	Endangered	Baker et al., 2008

^aMay no longer warrant this level of extinction risk; see Carlson et al., (2017).

2.3.3.2 Fishing methods

The most common capture method associated with inducing parturition was trawling, followed by longlines and gill-nets (Table 2.4). This difference may be due to a preference for the capture method used in the research, rather than induced parturition rate being influenced directly by fishing method. Another explanation for the greater rate of capture-induced parturition in trawl nets may be that parturition may be less likely observed when using methods such as gill nets or longlines since the loss of pups can occur at any time during the fishing process and neonates are not retained by these fishing techniques. Interestingly, it appears that more parturition events have been observed for rays than sharks (Table 2.4), and this is likely to be indicative of biases in catch composition for fishing methods. Globally, pelagic longline fisheries have the largest total annual shark bycatch, and deep-sea and coastal trawl fisheries have the largest total annual ray bycatch (Oliver et al., 2015). Two estimates for the frequency of capture-induced parturition caused by the same method (longlines) for *Prionace glauca* showed a discrepancy of just 3% (Table 2.2). On the other hand, estimates for both longlines and hook-and-line for *Pteroplatytrygon violacea* showed a discrepancy of 45% (Table 2.2). This provides some measure of the level of consistency of estimates within and across fishing methods and points to fishing method having some influence on the rate of capture-induced parturition. Further study is warranted to determine the

extent to which fishing methods may influence the occurrence and rate of capture-induced parturition.

Table 2.4 - The percentage of capture-induced parturitions observed with each fishing method differentiated into sharks and rays (n = 139 observations from 88 species).

Fishing method	Rays (%)	Sharks (%)	Total (%)
Artisanal fishing	0.7	0	0.7
Harpoon	0.7	0	0.7
Net (unspecified)	0	0.7	0.7
Gunshot	0	0.7	0.7
Tangle net	0.7	0	0.7
Hook-and-line	1.4	2.2	3.6
Seine-net	3.6	0	3.6
Gill-nets	3.6	8.7	11.6
Longlines	5.1	7.2	12.3
Trawling	12.3	4.3	16.7
Unknown or unspecified	8	11.6	19.6
Multiple	18.1	10.1	28.3
Total	54.2	45.5	99.7

Capture-induced parturition may occur at any time during the fishing process but is most often observed as animals are brought onto deck. Observation of capture-induced parturition prior to animals being brought on deck may be possible using gear mounted cameras. In addition, capture-induced parturition may be inferred from empty and distended uteri on capture, but would likely be an overestimate as some females may have recently given birth naturally. Parturition would be likely to be easiest to observe using capture techniques where the animal is hauled on deck relatively quickly and/or retained within a net. The mechanism causing parturition could be driven by interactions between a number of factors including oxygen deprivation, physical pressure due to the weight of other fish, lack of a support medium due to being removed from the water and stress caused by restraint or physical injury.

2.3.3.3 Reproductive mode

A major finding of this review is that the occurrence of capture-induced parturition is currently limited to live-bearing species, and there is no evidence that egg layers (skates, chimeras and some shark species) experience this condition. Within live-bearers, reproductive mode does not appear to influence the occurrence of capture-induced parturition. The proportion of species with capture-induced parturition was no different to that which would be expected if the 88 species were drawn randomly from all live-bearing species ($p = 0.640$) (Fig. 2.6).

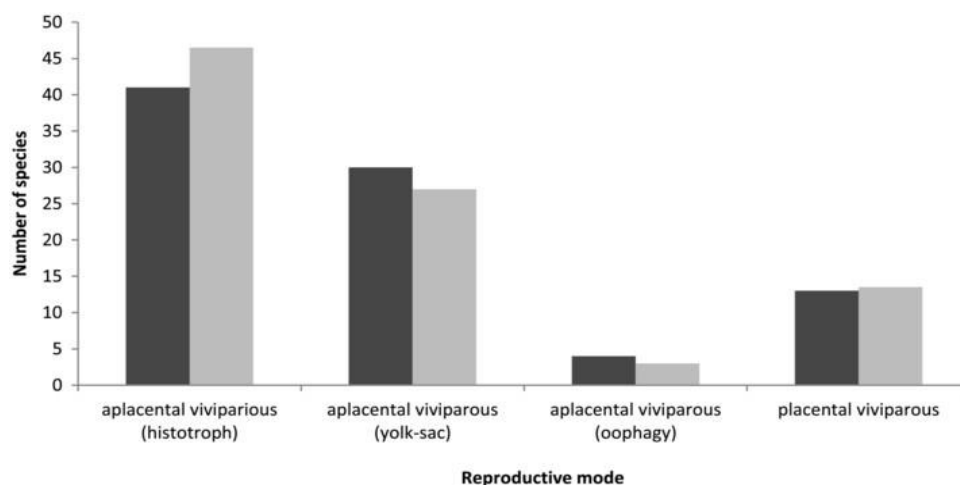


Figure 2.6 - The observed number of live-bearing species known to exhibit capture-induced parturition (black bars, $n = 88$ species) based on maternal provisioning compared to the number expected in each category calculated from the frequency of different reproductive modes of all live-bearing elasmobranchs (grey bars, estimated from Dulvy and Reynolds, 1997).

The consequence of capture-induced parturition for different reproductive modes is likely to differ for both the mother and the offspring. In terms of maternal input, matrotrophic viviparity through histotrophic uterine milk enables the female to gestate multiple, large offspring due to additional nutrient input (Grogan and Lund, 2011). They are likely to grow faster and have a greater birth-size than those supported through lecithotrophy because they are not limited to yolk sac nutrition (Grogan and Lund, 2011). This means that for matrotrophic species an abortion event is costlier to the female due to higher levels of maternal nutrient investment. On the other hand, matrotrophic offspring may be more likely to survive a capture-induced parturition at later stages of gestation due to their larger size.

Although observations of capture-induced parturition are currently limited to live bearing species, the possibility of such a response in egg-laying species remains. Three of the species “inferred” to show capture- induced parturition are egg-laying (see Adams, 2017). Port Jackson sharks (*Heterodontus portusjacksoni*) have also been observed to lay their eggs when handled (authors' pers. obs.). It is debatable whether laying in response to capture could be considered abortion in egg laying species, as the eggs are self-sustaining (Musick and Ellis, 2005). The majority of egg layers are sequential bearers with one egg deposited at a time from each oviduct (Musick and Ellis, 2005), and tens to hundreds of eggs may be deposited over a season (Musick and Ellis, 2005). Also, eggs are laid frequently, with some species such as *Raja clavata* laying every 24 h (Holden, 1975). Therefore, if an egg is released due to capture, it is likely that the egg would have been laid naturally in the near future anyway. One potential issue with releasing eggs on capture is that species such as catsharks (Fa. Scyliorhinidae) secure their egg cases to algae or rock (Smith and Griffiths, 1997). Such eggs may be expected to have a high mortality rate if released on deck and returned loose to the water. Additionally, in oviparous species with ‘multiple oviparity’, embryos in the egg cases begin to develop inside the mother's body. Usually an egg case is laid when the embryo in it grows to a certain length (Nakaya, 1975). In these species capture-induced laying could result in a reduction of the period inside the mothers' body and therefore an increased rate of mortality.

2.3.3.4 Gestation stage

The gestation extent of capture-induced embryos ranged from early-stage eggs to fully-formed near-term pups. Eggs were aborted by ten live-bearing species in total with the remaining species giving birth to early, mid and near-term embryos (Adams, 2017). No early or mid-term embryos were reported to be birthed by placental viviparous species, possibly because their placental connection may physically reduce the chance of capture- induced parturition. This placental

connection, however, only forms part way through gestation; for example, embryos of the smooth dogfish (*Mustelus canis*) develop a yolk-sac placenta about three months into their 10 to 11 month gestation period (Price and Daiber, 1967). Before the placental connection forms, the free-floating embryo could still be capture-induced. At this stage, very little is known about embryo survival, however Charvet-Almeida et al. (2005) observed that the embryos of freshwater stingrays (*Potamotrygon* spp.) hardly ever survived after capture-induced parturition regardless of their developmental stage. Whether this can be considered a rule for this group of species, and for live-bearing elasmobranchs in general, remains undetermined. At least ten live birthing species are also known to have aborted eggs upon capture. Due to their early developmental stage, these would have had no chance of survival. Future studies could assess the post-capture survival of females and capture-induced pups to determine the chance of survival for near-term embryos. The swimming speed of neonates and their feeding ability could also be affected by a premature parturition event and any influence on long-term survivorship and fitness of embryos should be investigated. A lack of standardised terminology was noted when reporting the gestation extent of capture-induced embryos. Future studies should report the presence and size of yolk-sacks and whether embryos appear early, mid or near-term.

2.3.4 Social media as a source of useful information and misinformation

In total, 40 videos and one image series were identified on social media sites showing capture-induced parturition in sharks and rays (Table A1.1; Table A1.2; Table A1.3; Table A1.4). In many cases, these videos provide visual documentation that confirm the observations of capture-induced parturition in the literature. Species identification was not possible in many cases as the geographic location was not known and the quality of the footage poor. The majority (61%) of the species in the videos were caught by recreational fishers using hook-and-line, with the remainder caught by researchers using longlines (3%) or strike nets (3%) or with unknown fishing methods (33%). In terms of taxonomy, 12% of the videos show sharks and 88% batoids which may represent a bias in capture not mirrored in the observations identified in the scientific literature. We presume that the uploaders of these videos were unaware of the true nature of the event. This is supported by the optimistic titles of many of these videos: “Man catches stingray while it’s giving birth..unhooked and realaesed [sic]...!” (> 1.7 million views); “Caught On Camera: fisherman helps stingray give birth” (> 19 million views and with the hashtag “happybirthray”) (Table A1.2). The fishers are often seen to intervene and remove the offspring, seemingly thinking they are assisting the animal. Such representation feeds into the narrative that these births are a spontaneous occurrence rather than an event which is caused by capture.

Shark species identifiable in the videos include a lemon shark (*Negaprion brevirostris*) and a longfin mako (*Isurus paucus*) which may have been dead at the time of the video, but in the literature has been observed giving birth after capture (Gilmore, 1983). The video of the lemon shark provides evidence that this species gives birth in response to stress. Notably the individual in the video was tagged with an internal acoustic transmitter prior to release. Nine readily identifiable ray species exhibit capture-induced parturition on camera (Table A1.1). These include four additional species not observed in the literature: the critically endangered smalltooth sawfish (*Pristis pectinata*), the endangered giant freshwater stingray (*Urogymnus polylepis*), the lesser guitarfish (*Rhinobatos annulatus*) and the white-spotted whiptail (*Maculabatis gerrardi*). Videos confirming capture-induced parturition in ray species already observed in the literature include the white-spotted eagle ray (*Aetobatus narinari*), the estuary stingray (*Hemirhamphysa fluviorum*), the bat ray (*Myliobatis californica*), the Atlantic stingray (*Hypanus sabinus*) and the bluntnose stingray (*Hypanus say*). Interestingly, photos from Lüderitz Marine Research show a likely abortion event caused by stranding in the lesser guitarfish (*R. annulatus*). If those four additional ray species (*P. pectinata*, *U. polylepis*, *R. annulatus* and *M. gerrardi*) and one additional shark species (*N. brevirostris*) found via social media are included in our estimate of capture-induced parturition, it brings the total species count to 93 and the endangered species count to 9.

2.3.5 Reducing the likelihood of capture-induced parturition

Until we understand the specific mechanisms that induce parturition on capture, general techniques to reduce stress should be encouraged for scientists and fishers. Cooke and Suski (2005) identify certain handling techniques which can significantly reduce stress and post-release mortality in fish. These general techniques which can be adopted to reduce the impact of fishing are (1) minimising angling duration, (2) minimising air exposure, (3) avoiding angling during extremes in water temperature, (4) use of barbless hooks and artificial lures/flyes, and (5) avoiding angling fish during reproductive periods (Cooke and Suski, 2005). For researchers conducting studies on sharks and rays listed above (and more broadly) we would recommend conducting procedures without removing the animal from the water, especially for larger shark and ray species. For endangered species, it would seem logical to avoid sampling in periods or areas where females are pregnant, or use selective fishing techniques so pregnant females can be avoided. We need research to quantitatively assess optimal approaches to reduce capture-induced parturition. Further, a better understanding of the mechanisms of parturition should provide clear

guidance on mitigating capture-induced parturition; however, measures outlined above appear to be logical first steps.

2.3.6 Management strategies, recommendations and concluding remarks

The above synthesis demonstrates the prevalence of capture-induced parturition by live-bearing elasmobranchs in response to various methods of capture. Although capture-induced parturition does not appear to correlate with higher threat levels, it represents a potentially threatening process that is rarely considered within population or fisheries models and threat assessments. This is especially telling as species with high rates of post-release survival are currently considered to be largely unaffected, despite potentially losing considerable reproductive potential for that cycle (the whole reproductive output for up to two years for some species). It is possible that the condition may affect recruitment for a substantial proportion of live-bearing sharks and rays. Clearly, immediate research is required to determine the magnitude of effect on these populations, with focus on threatened species (Table 2.3).

The data used in this review provides the first list of species known to exhibit capture-induced parturition (Adams, 2017) including a number of elasmobranchs that are highly threatened (Table 2.3). We recommend that future sampling techniques for such species should be carefully considered, given the likelihood of many common sampling methods causing capture-induced parturition. There is the potential for a large number of currently data deficient species to also exhibit capture-induced parturition, which is concerning given their lack of threat assessment.

Considering it is the stress associated with capture (irrespective of whether the animal is released after capture) that is the source of the problem, the only means of mitigation are likely to be seasonal and/or spatial closures designed to protect species while they are pregnant (especially for those species that are endangered). Internationally, there is a growing body of evidence supporting hypotheses that no-take marine reserves help conserve some shark populations (Meekan and Cappel, 2004; Garla et al., 2006; Heupel et al., 2009; Bond et al., 2012; White et al., 2017). Acting in an analogous fashion to spatial closures, seasonal closures protecting nursery areas or aggregation sites offer a temporary spatial refuge for affected species during critical reproductive periods. An example is the protection of the school shark (*Galeorhinus galeus*) through seasonal closures in certain locations off southern Australia during the pupping season (Bensley et al., 2010). Simpfendorfer (1999) and Prince (2005) suggest that targeting protection of breeding size adults is an important management strategy for chondrichthyan fisheries, rather

than juveniles and sub-adults that are conventional targets of fisheries management. This strategy would allow pregnant females to give birth to full-term pups and would contribute to healthy levels of recruitment.

No-take marine reserves have been predicted to influence surrounding biodiversity due to the ‘spill over’ of adults and juveniles across borders (Botsford et al., 2003; Gell and Roberts, 2003) thereby replenishing fished areas outside the Marine Reserve (Roberts et al., 2001). Similarly, for species that show capture-induced parturition, permanent marine reserves may represent an important source of juveniles. To be effective, no-take marine reserves need to be designed around the critical life stages of multiple species, and work by protecting the habitats on which these stages depend (Bonfil, 1999). Nursery area protection is important for shark and ray management as these nearshore areas are often intensely used by humans. Further research is needed to develop management strategies that encompass older individuals residing outside nurseries (Kinney and Simpfendorfer, 2009), especially those large females that are susceptible to capture-induced parturition due to fishing.

The posting and sharing of videos by both members of the public and scientific organisations which unconsciously depict capture-induced parturition events, highlights a lack of knowledge regarding this response to stress. There may be some benefit in a broader communication strategy which highlights the risk that capture of pregnant elasmobranchs can cause premature birth or abortion. It would also be useful to assess if certain post-capture handling techniques can reduce capture-stress and associated parturition for elasmobranchs. In addition, it would be useful to quantify actual mortality of embryos following capture-induced parturition, to better quantify the magnitude of this problem. With better understanding of the physical and physiological mechanisms which induce parturition, it may be possible to develop techniques to reduce stress-induced parturition rates on vulnerable species, and thereby reduce the impact of catch and release angling, or fishing methods where adult elasmobranchs are discarded. At any rate, it is clear that resource managers need to consider the indirect threats to elasmobranchs posed by fishing.

2.4 References

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Chapter 3 – Movement and space use of a demersal elasmobranch species, *Trygonorrhina fasciata*, in relation to marine protected areas

3.0 Abstract:

Populations of sharks and rays (elasmobranchs) are impacted by fishing activities, with ray species (batoids) identified as those being the most threatened. Marine protected areas are one means to potentially mitigate human impact by providing areas where fishing and other impactful activities are restricted. The success of marine protected areas depends largely on the rate and scale of animal movements, however the movement and space use of rays is poorly understood, and therefore the applicability of spatial closures to fishing (marine protected areas) to management of batoids requires targeted study. This is particularly relevant in soft-sediment habitats, as it is often assumed that species in this homogenous environment have little reason to show residency or site attachment and are therefore wide-ranging. We used acoustic telemetry to determine long-term residency and a fine-scale receiver array to determine the space use of eastern fiddler rays *Trygonorrhina fasciata*. This species showed substantial residency to soft-sediment areas, with 71% of animals repeatedly using the same areas for extended periods. The movement patterns of *T. fasciata* varied by gender and sexual maturity, with females having higher residency but using larger areas than males. Further, immature rays showed higher residency than mature individuals. The majority of rays displayed restricted patterns of movement which is conducive to protection by spatial closures to fishing. We provide data that suggests no-take areas can be effective management tools for resident batoids, particularly in near-shore soft-sediment habitats. These findings are relevant to the conservation and management of batoid species and provide evidence that soft-sediment species are not all wide-ranging.

3.1 Introduction:

There is increasing evidence of the effects of fishing on abundance, size/age structure and population dynamics of sharks and rays (elasmobranchs) (Stevens, 2000; Dulvy et al., 2008). Declines of many elasmobranch species have been documented in fisheries around the world, attributable to targeted elasmobranch fisheries and fisheries where elasmobranchs are caught as bycatch (Stevens, 2000; Dulvy et al., 2008; Dulvy et al., 2014). In particular, fishing is of concern for ray species (batoids) which are frequently caught in fisheries but are often misidentified, aggregated in catch records, or discarded, with catches in many parts of the world being unregulated (Dulvy et al., 2014). This presents difficulties for monitoring and management (Clarke et al., 2006; Bornatowski et al., 2013; Dulvy et al., 2014). Globally, chondrichthyan catch has been increasingly dominated by batoids. This group has the highest global species richness among chondrichthyans (51.8% of the ~1041 species worldwide) and consistently make up >

50% of taxonomically differentiated global landings (Dulvy et al., 2014). Five out of the seven most threatened families of chondrichthyans are batoids with some of the most threatened having declined due to incidental capture (Dulvy et al., 2014). Bycatch and discarding causes impacts ranging from post-capture mortality to reduced fitness and reproductive output as a result of stress and injury (Guida et al., 2017; Rodrigues et al., 2018; Wosnick et al., 2018). These potential impacts are difficult to manage because they depend on the fishing method used and on the species involved. The issue of bycatch is of particular concern given the magnitude of the catch, and the documented vulnerability of the species involved. Marine protected areas are one means to potentially mitigate this impact by providing areas where fishing activities are restricted.

Marine protected areas can show positive effects on species (Roberts et al., 2000; Gell and Roberts, 2003; Botsford et al., 2010), but the magnitude and extent of most effects depend largely on the rate and scale of animal movements (Kramer and Chapman, 1999). Space use is variable for sharks and rays, and therefore the applicability of marine protected areas to management of these species requires targeted study. To ensure the design of MPAs is adequate to protect elasmobranchs, a detailed understanding of their movement patterns and spatial requirements is needed (Chapman et al., 2005; Block et al., 2011; Papastamatiou and Lowe, 2012). However, when establishing MPAs or networks of MPAs, there is often little or no data on the movement of the species these reserves are designed to protect. This is notably the case for batoid species; we lack information about their populations and movements despite the high level of fishing impacts and disproportionately high number of threatened species (Dulvy et al., 2014; Heupel et al., 2019). The movement behaviour of elasmobranch species has been shown to be highly variable on both spatial and temporal scales (Heupel et al., 2004). Highly mobile or migratory species may move in and out of protected areas frequently and as a result MPAs, or individual no-take marine reserves, may be too small to protect such species. However, many small-bodied shark species spend their entire life in nearshore coastal habitats (Heupel et al., 2019) and there is growing evidence of resident or philopatric behaviour in various batoid taxa (Dewar et al., 2008; Wearmouth and Sims, 2009; Hunter et al., 2006; Walker et al., 1997). High levels of residency hint that some species may indeed benefit from the protection of no-take zones (Wearmouth and Sims, 2009) and indeed a number of elasmobranch species have been shown to respond to protection (Meekan and Cappo 2004; Heupel et al., 2009) or move on spatial scales relevant to the scale of current protected areas (Hueter et al., 2005; Garla et al., 2006; Heupel et al., 2010; Escalle et al., 2015; Acuña-Marrero et al., 2017). Some species show variation in movement behaviour as they transition through different life-history stages (Knip et al., 2010).

Therefore, knowledge of space use and the identification of key habitats for specific life-history stages, would contribute greatly to the effectiveness of the design of MPA networks (Heupel and Simpfendorfer, 2005; Block et al., 2011; Simpfendorfer et al., 2011; Papastamatiou and Lowe, 2012; Heupel et al., 2019).

Animal movements and behaviour patterns are driven by abiotic cues such as water temperature along with habitat type with these factors limiting species distributions and triggering behavioural responses (Lowe and Bray, 2006). Water temperature has been shown to influence the behaviour, ecology and movements of elasmobranchs (Snelson and Williams, 1981; Matern et al., 2000; Hopkins and Cech, 2003; Schlaff et al., 2014). These effects vary with species and depend on their physiology and habitat requirements (Vaudo and Lowe 2006). To be effective, no-take marine reserves need to be designed around the interaction of animals and their critical habitats (Heupel et al., 2007). Shallow coastal marine environments such as estuaries and embayments are known to provide key habitats for many shark and ray species (Heupel et al., 2007). Due to their proximity to human population centres, these regions are increasingly impacted by pollution, development, and recreational use (Knip et al., 2010). These shallow nearshore areas are also the focus of a large amount of fishing effort (Stewart et al., 2010, Knip et al., 2010). Subsequently it is in these areas that threat levels of shark and ray species to fishing are highest (Stewart et al., 2010, Dulvy et al., 2014). For the species that use these areas, it means that individuals may stand a high chance of being caught, interacting with fishing gear and experiencing post-capture mortality or aborting their offspring (Chapter 2; Adams et al., 2018). It is therefore important to identify high use areas and habitats, as well as understand environmental factors driving behaviour to inform spatial management and maximise protection. A specific habitat used by many batoids has often been overlooked in the design of MPAs despite its global prevalence; soft sediments.

It is often assumed that species utilising soft sediment habitats have little reason to show site attachment and are therefore wide ranging (Caveen et al., 2012). This assumption is made despite the limited data available on the movement behaviour of demersal fish in marine soft sediment habitats, particularly in relation to MPAs (Fetterplace et al., 2016, Fetterplace 2018). The assumption of broadscale movements in this habitat tends to be based on their homogeneous appearance and lack of structure, with little reason for fish to be resident (Lowe and Bray 2006, Caveen et al., 2012, Fetterplace et al., 2016). Recent work has challenged this assumption with evidence building of residency on soft sediments for some fish species (Fetterplace et al., 2016). However, residency and space use of larger bodied fish species, including batoids, in this habitat remains poorly resolved. This is particularly relevant to spatial

management, with recent changes in some Australian MPAs resulting in the opening of some no-take zones to recreational fishing (Brooks et al., 2013). These changes were influenced by the belief that fish on soft sediments are not resident and therefore the no-take areas on soft sediments provide little conservation value (Fetterplace 2018).

The aim of this study was to examine the long-term residency and fine-scale space use of eastern fiddler rays *Trygonorrhina fasciata* Müller and Henle 1841 to determine how they use habitat and whether they move on spatial scales relevant to current marine protected areas. Passive acoustic telemetry was used to measure the residency and determine the space use of this species over time to assess the level of protection afforded to them by no-take areas. My first hypothesis was that *Trygonorrhina fasciata* would show residency to areas able to be encompassed by the size of current no-take areas and this level of residency may differ with sex and life-history stage. My second hypothesis was that there would be differences in space use among individuals based on sex, maturity stage and water temperature. We contextualise our findings by examining space use and movement patterns in this batoid species relative to other batoid taxa species.

3.2 Methods

3.2.1 Study site

Jervis Bay is a shallow embayment on the southeast coast of NSW, Australia (Fig. 3.1) with most of the Bay being less than 10 m deep and a maximum tidal range of 2 m. The Bay contains a range of habitat types, but is dominated by soft sediment habitat interspersed with a range of other habitat types including rocky intertidal and subtidal reefs along with seagrass beds around the Bay's perimeter (Fetterplace, 2018). The area falls within the bounds of Jervis Bay Marine Park (JBMP) which is a network of no-take marine reserves (sanctuary zones) and habitat protection areas aimed at conserving biodiversity (MPA, 2009). We conducted this study in two no-take sanctuary zone sites within Jervis Bay (Fig. 3.1a, b) and one fished site on the open coast (Fig. 3.1c). Thirty kilometres to the south of Jervis Bay, is the third site; Bendalong. Fishing is permitted at Bendalong and the site is characterised by similar habitat features as Jervis Bay but lacks seagrass beds and is more exposed to waves. We deliberately chose Bendalong as a comparison site because it was on the open coast and therefore different from the sites inside the bay in terms of wave energy. This was to provide movement data in an area that was not within a sheltered embayment.

The NSW Department of Primary Industries and collaborators (University of Wollongong, Integrated Marine Observing System and Macquarie University) maintain an array of acoustic

receivers (Vemco Ltd., Canada) in Jervis Bay and Bendalong to monitor the movement of marine animals. Receivers have been deployed over several years; 49 receivers were deployed in 2015, and a further 10 were deployed at Bendalong in 2018. Data were downloaded from receivers approximately every 12 months in Jervis Bay and after 8 months at Bendalong.

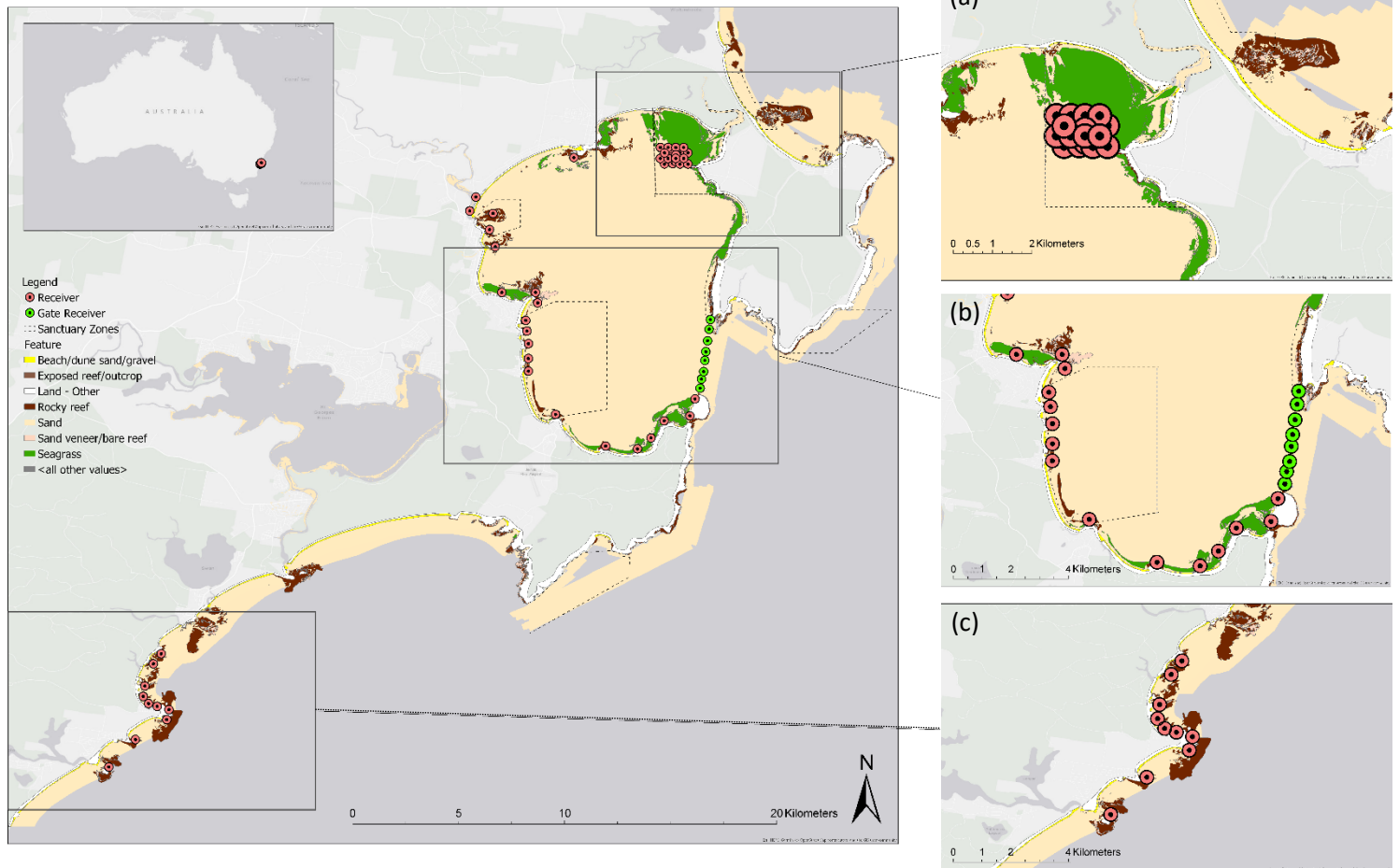


Figure 3.1- The acoustic receiver network in Jervis Bay and Bendalong with three sites targeted in this study; (a) Hare Bay no-take sanctuary zone with a fine-scale positioning array (VPS) (b) Hyams no-take sanctuary zone and (c) Bendalong receiver array.

3.2.2 Study species

Long-term acoustic tracking data were collected for eastern fiddler rays *Trygonorrhina fasciata*, a species endemic to south eastern Australia. Banjo rays (Trygonorrhinidae) are large rays which are strongly associated with soft-bottom habitats in shallow (< 100 m) coastal waters, resulting in potentially high exposure to fisheries (Knip et al., 2010, D’Alberto et al., 2019). *Trygonorrhina fasciata* is a ubiquitous species endemic to eastern Australia between southern QLD and Twofold Bay, NSW. It has been observed in Jervis Bay on reefs, in the shallow subtidal, in seagrass and over soft sediment areas (CSIRO, 1994; Broad et al., 2010; Kiggins 2013; Rees, 2017). The benthic habit and apparent preference for nearshore, shallow-water habitats predisposes this species to capture by a variety of fishing methods (Dulvy et al., 2014). There is limited data on movements for this species, but tracking of 9 males on an artificial reef off Sydney suggests they may show affinity to the site, at least in the short term (Keller et al., 2017). Space use remains unstudied for this species and species group.

3.2.3 Field methods

Thirty-five eastern fiddler rays *Trygonorrhina fasciata*, were captured at three sites using rod and line with baited circle hooks or hand captured by freedivers (at Bendalong) and surgically implanted with acoustic tags using conventional methods (Barker et al., 2009). Tagging occurred in stages over a period of up to 4 years and therefore all individuals were not tracked simultaneously. Within Jervis Bay, 14 females and 11 males ranging from 580 – 920 mm in total length were caught and tagged in Hare Bay no-take sanctuary zone within range of a 20 Receiver fine-scale receiver array (VPS) enabling high resolution positioning (Fig. 3.1a). Range testing indicated V9 transmitters had a maximum detection range of 420 m based on 5% probability of detection and the distance at which 50% of transmissions were detected was ~250 m (Fetterplace 2018). Rays at this site were tagged in three batches; in Autumn 2015 (n = 6), Summer 2015 (n = 6) and Autumn 2017 (n = 13). At the second site, Hyams no-take sanctuary zone, 3 females and 2 males ranging from 705 – 765 mm in total length were caught and tagged in December 2015 (Fig. 3.1b). At the third site, Bendalong, 3 females and 2 males ranging from 715 – 955 mm in total length were caught and tagged in August 2018 (Fig. 3.1c).

After capture, rays were held in 50 L holding tanks and anaesthetized in seawater containing 60 mg L⁻¹ of Aqui-S. Once unresponsive, the fish were measured (TL) and a mid-ventral incision was performed after disinfection of the incision site and surgical equipment using surgical grade iodine. A uniquely coded V9 or V13 acoustic transmitter (for tag details Table A2.1) was inserted through a mid-ventral incision in the abdomen. The incision was closed with two or three

dissolvable stitches tied with a double surgeon's knot. Rays were then transferred to a holding tank and monitored for recovery, before release at the site of capture. All relevant DPI permits (Permit No: P15/0003-1.0) and UOW ethics approval (Ethics Number: AE14/25) were obtained for this study.

3.2.4 Residency Analysis

Residency of fiddler rays within the Hare Bay no-take sanctuary zone was analysed using a residency index (RI). RI was defined as the number of days a tagged fish was detected, divided by the number of days monitored (Garcia et al., 2015). Days monitored ended either when the study ended or when the tag battery expired, whichever occurred first. This resulted in a monitoring period between 155 and 904 days. A generalized linear mixed-effects model with a binomial distribution was used to test the effect of sex and stage of maturity on residency. To account for variation caused by tagging site, site was included in the model as a random factor.

3.2.5 Space Use Analysis

Individual positions were estimated for individual rays by Vemco (Vemco Ltd., Canada) using their proprietary hyperbolic positioning algorithms (Smith, 2016). I determined monthly activity spaces of rays using 95% and 50% kernel utilization distributions (KUDs) (Worton, 1989). Kernel analyses involve the construction of a probability density function around each animal position. The shape and width of the kernel was determined by a smoothing parameter or bandwidth, denoted by h (Worton, 1989). An adaptive *ad hoc* method for a bivariate normal kernel was used to select smoothing parameters for kernel estimation (Worton, 1989; Calenge, 2006). All KUD analysis and mapping was conducted using the R package *adehabitatHR* (Calenge, 2006) on the Zoatrack platform (Dwyer et al., 2015). KUDs made up of fewer than 100 points in a given month were excluded from analyses as they were highly variable and I did not consider them reliable estimates of animal behaviour (Fig. A2.1). Space use patterns for transitory rays were not considered in KUD analysis, because the movement and space use patterns of these individuals may not be indicative of local space use and reliable positioning of these animals was poor. Therefore, space use analysis was restricted to resident individuals that were present for more than 6 months.

A linear mixed effects model was used to test the effect of sex, maturity and water temperature on monthly 50% and 95% KUD size. Water temperature was measured using Hobo data loggers (OneTemp Pty. Ltd.) and included in the model as average monthly temperatures (Fig. A2.2). The total length of rays had a bimodal distribution and therefore was analysed as a categorical

factor with two levels (immature, mature) to estimate its effect on activity space. Individual tag ID was included as a random factor to account for the repeated measures in these data. Log x +1 transformation was used to achieve homogeneous variances. Models were computed using the *nlme* package in R (Pinheiro et al., 2012). Models were compared using Akaike's information criterion with a small sample size bias correction (AICc).

3.3 Results

3.3.1 Residency

Among the 35 fiddler rays tracked, movement patterns could broadly be divided into four distinct categories:

- (1) long term, essentially continuous residency for the entire monitoring period (n = 17, 48.6%, Fig. 3.2)
- (2) continuous residency for multiple months followed by an abrupt absence (n = 8, 22.9%, Fig. 3.2)
- (3) repeat site attachment for short periods (weeks to months) interspersed with long periods of absence and then reappearance (n = 4, 11.4%, Fig. 3.2)
- (4) left the tagging location almost immediately after tagging and did not return (n = 6, 17.1%, Fig. 3.2 with n=5 detected leaving Jarvis Bay)

Five individuals from Hare Bay were also detected on the gate receivers across Jarvis Bay entrance, with most only detected for a short period, likely leaving the bay. One female (A69-1601-10875, 890 mm TL) was detected leaving and returning to Jarvis Bay on multiple occasions, showing residency to Hare Bay each summer and then leaving Jarvis Bay for the remainder of the year. The maximum distance moved by an individual was ~3750 km (based on 126103 positions over 20 months) all within a 1.12 km² area. It should be noted that distance moved is correlated to the number of positions so presenting summary statistics on distance moved beyond the maximum is not particularly meaningful.

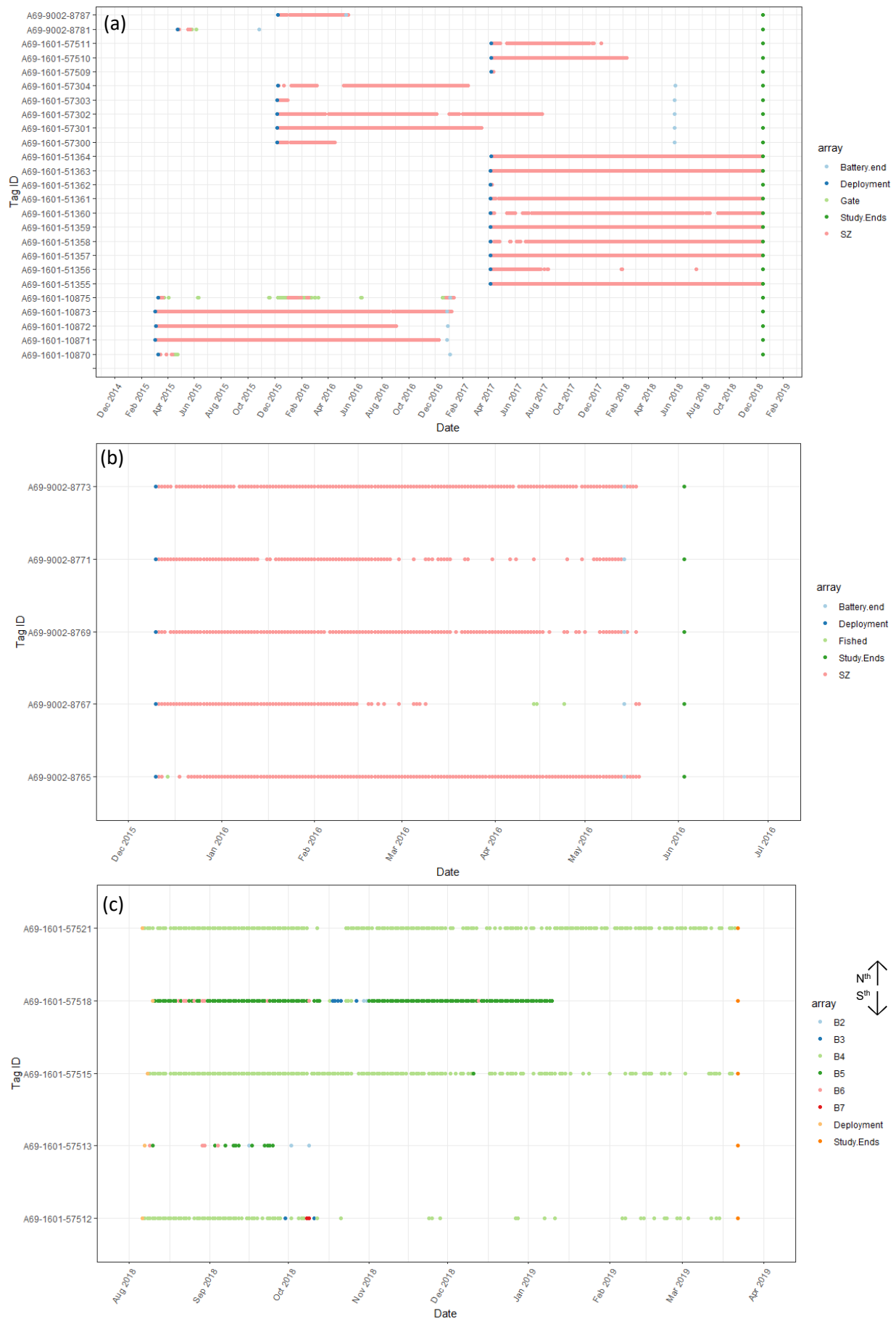


Figure 3.2- Detection plots showing the daily presence/absence of 35 eastern fiddler rays *Trygonorrhina fasciata*, to three sites a) Hare Bay (n = 25), b) Hyams (n = 5) and c) Bendalong (n = 5). Individuals at sites (a) and (b) were tagged within a no-take sanctuary zone (SZ) and detections in fished areas outside this zone are indicated. Site (c) is a fished area but receivers are deployed on a similar spatial scale to those deployed within sanctuary zones in sites (a) and (b).

Average residency to the tagging sites was substantial (> 50%) but changed with sex and stage of maturity. Females on average showed significantly higher residency (0.720 ± 0.081 SE) than males (0.560 ± 0.096) (Estimate = -0.70300 ± 0.03083, $z = -22.799$, $p < 0.0001$) (Fig. 3.3a). Immature rays also showed higher residency (0.733 ± 0.077) than mature individuals (0.523 ± 0.096) (Estimate = -0.91732 ± 0.03160, $z = -29.029$, $p < 0.0001$) (Fig. 3.3b).

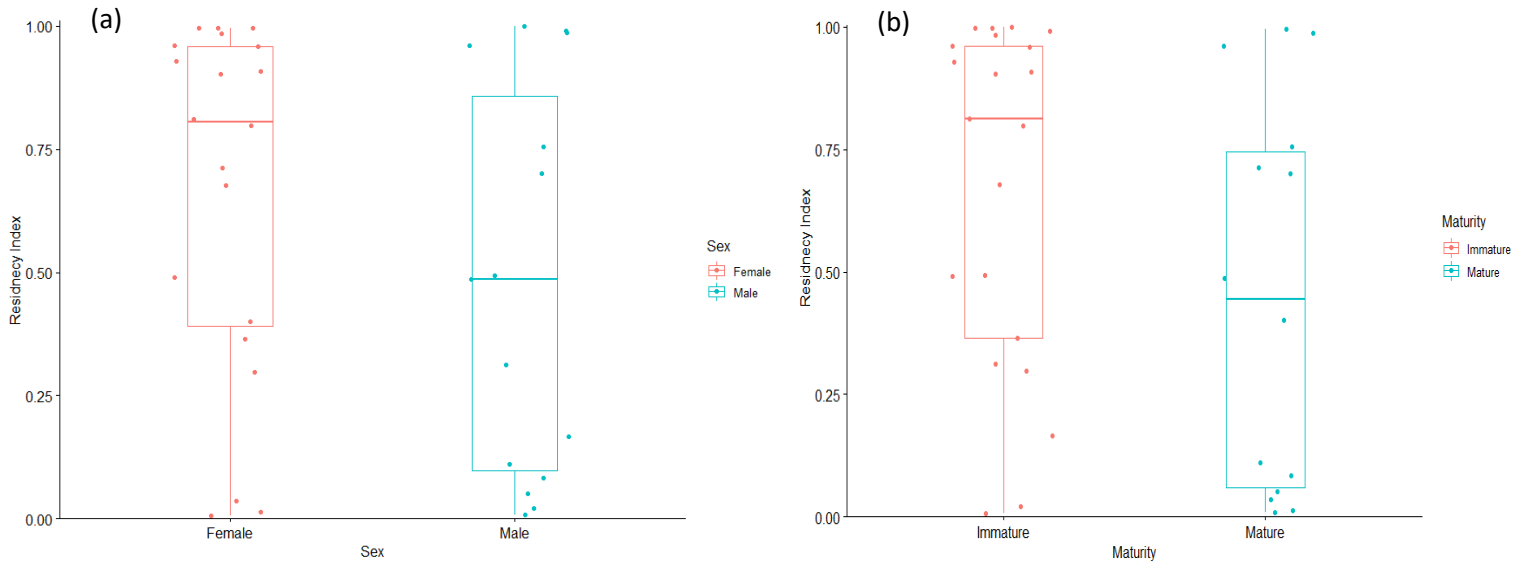


Figure 3.3- The effect of (a) sex and (b) maturity on the residency index (days detected/days monitored) for *Trygonorrhina fasciata* (VPS) (n=35), with median shown as a horizontal line.

3.3.2 Space Use

Activity space size varied substantially between sexes and with water temperature (Table 3.1). Sex had a significant effect on monthly 95% ($F_{(1,10)} = 8.393$, $p = 0.0159$) KUD size with females using larger areas than males (Fig 3.4a, Fig. 3.5). The 50% KUD size showed the same trend (Fig. 3.4b) but the difference was not significant ($F_{(1,10)} = 3.299$, $p = 0.0994$). Temperature had a significant effect on both 50 % ($F_{(1,166)} = 7.260$, $p = 0.0078$) and 95% ($F_{(1,166)} = 9.656$, $p = 0.0022$) monthly KUD size, with smaller KUDs associated with higher water temperatures (Fig. 3.6).

Table 3.1- Effect of sex, maturity and water temperature on monthly space use of *Trygonorrhina fasciata* (50% and 95% kernel utilisation distributions).

Model	df	95% KUD AICc	50% KUD AICc	95% KUD Δ AICc	50% KUD Δ AICc	95% KUD Deviance	50% KUD Deviance
KUD ~1	3	150.719	249.2341	12.8461	12.6406	-144.59064	-243.105
KUD~ Sex + Maturity + Temp	6	139.9585	237.1108	2.0856	0.5173	-127.47018	-224.6224
KUD~ Sex + Maturity	5	149.4112	249.2341	11.5383	12.6406	-139.08514	-238.908
KUD~ Sex + Temp	5	137.8729	236.5935	0	0	-127.52608	-226.2468
KUD~ Temp + Maturity	5	143.1784	239.0814	5.3055	2.4879	-132.83156	-228.7346
KUD~ Temp	4	141.502	237.1344	3.6291	0.5409	-133.27216	-228.9046
KUD~ Maturity	4	152.2203	251.1221	14.3474	14.5286	-144.00412	-242.9058
KUD~ Sex	4	147.3315	248.3212	9.4586	11.7277	-139.11524	-240.105

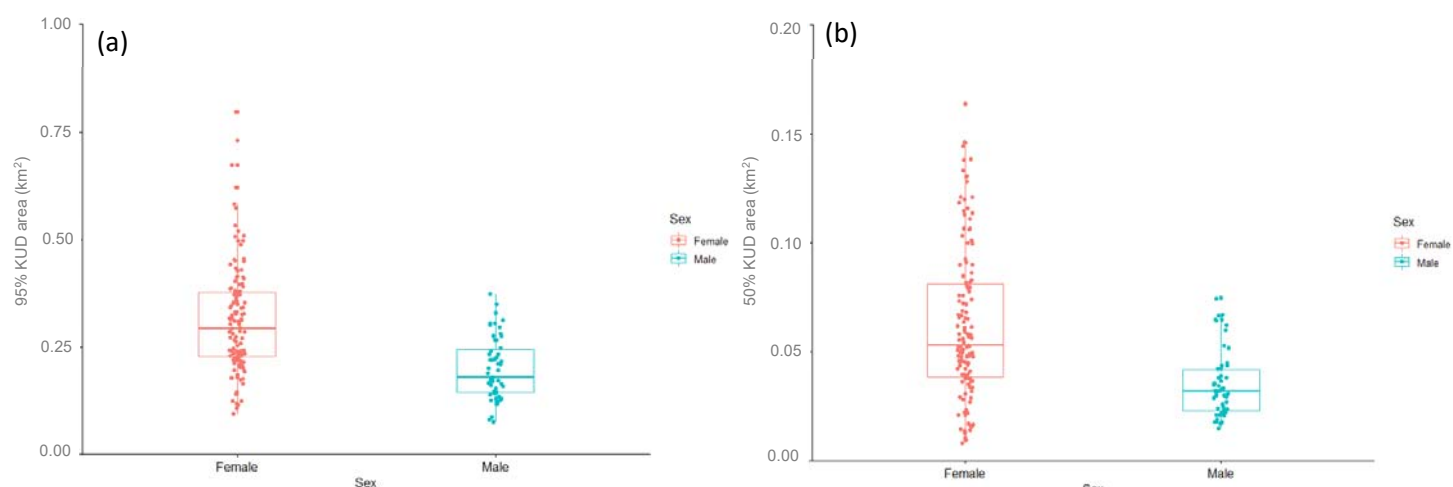


Figure 3.4 - The effect of sex on monthly space use (a) 95% and (b) 50% KUDs for *Trygonorrhina fasciata* as determined via positioning with a fine-scale acoustic receiver array (VPS) (n=179), with median shown as a horizontal line.



Figure 3.5 - Illustrative 50% (inner) and 95% (outer) space use KUDs for one month of tracking of 9 female (red) and 3 male (blue) *Trygonorrhina fasciata* in a fine-scale acoustic receiver array (VPS). The 20 VPS receiver positions are indicated with red points.

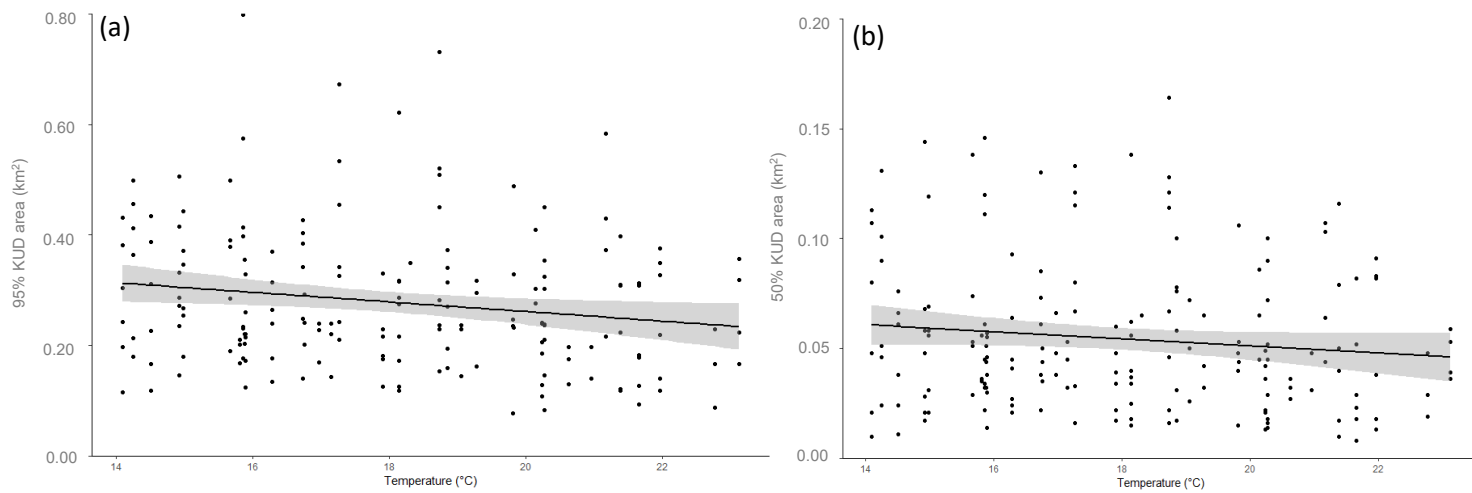


Figure 3.6- the effect of water temperature on the size of monthly space use (a) 95% and (b) 50% KUDs for *Trygonorrhina fasciata* as determined via positioning with a fine-scale acoustic receiver array (VPS) (n = 179).

3.4 Discussion

Here I show a demersal batoid species *Trygonorrhina fasciata* show high levels of residency to areas of soft-sediment for long periods, which elevates their chance of protection. This adds to growing evidence that Marine Protected Areas can be effective management tools for elasmobranch population as long as the spatial and temporal dynamics of species are considered when designing them (Port et al., 2012). This is especially important nearshore areas, which are highly productive but experience high impacts (Knip et al. 2010). Previous research has revealed nearshore habitats such as mangroves, seagrass beds and reefs and can be important nursey areas and provide key habitat for a range of species (Heupel et al., 2007; Vaudo and Heithaus, 2012; Cerutti- Pereyra et al., 2014; Davy et al., 2015), The current study expands this to include soft-sediment habitats, with rays showing high residency to small areas for extended periods.

Residency to the tagging sites was substantial and the area of space use was small, with resident animals repeatedly using the same areas for extended periods. This is despite *T. fasciata* having a demonstrably high capacity for movement, with the highest measured cumulative distance moved equivalent to the entire east coast of Australia, however this amount of ground was covered within a 1.12 km² area inside a no-take zone. This demonstrates strong site attachment to soft-sediment habitat, confirming work by Fetterplace et al. (2016) on smaller teleost fish species, and challenging previous speculation (Caveen et al., 2012) regarding a lack of site-attachment over the long term. Site attachment in other nearshore habitats has also been shown for the juveniles of other ray species (e.g. Vaudo and Heithaus, 2012; Cerutti- Pereyra et al., 2014;

Davy et al., 2015), suggesting that no-take areas in nearshore environments can assist in maintaining some batoid populations (Davy et al., 2015). The movement patterns of *T. fasciata* also fall within the area encompassed by current no-take zones, which indicates that Jervis Bay Marine Park has zones that are adequately sized and placed to protect this species. This cumulative sanctuary zone size in Jervis Bay (4,300 ha) is the smallest in NSW with the other five marine parks having larger total areas protected (Solitary Islands: 8900 ha, Lord Howe: 12800 ha, Cape Byron: 6100 ha, Port Stephens: 19,700 ha, Batemans: 17,000)(Read, 2014). These findings are consistent with MPAs providing protection for resident batoid species which has particular relevance for locations with high levels of impact which are used during critical life stages.

The movement patterns of *T. fasciata* varied among individuals, with not all rays exhibiting the same behaviours. Such variation in behaviour is not surprising; there is variation in individual traits driven by morphological or physiological attributes that can lead to differences in movement behaviour (Clobert et al., 2009). Through ontogenetic changes, different individuals also have different requirements based on their life-stage and attributes such as size and sex which can manifest in differences in movement behaviour (Speed et al., 2010). The movements of various ray species have previously been linked to reproduction (Gray et al., 1997; Hoisington and Lowe, 2005; Vaudo and Lowe, 2006, Port et al., 2012), and seasonal changes in water temperature (Hoisington and Lowe, 2005; Vaudo and Lowe, 2006; Dewar et al., 2008; Vaudo and Heithaus, 2009; Port et al., 2012). The differences in residency and space use I observed in *T. fasciata* are also likely linked to these factors. Presumably females exhibited larger areas of space use due to the greater energy requirements associated with their larger body size and possibly gestation; hence the need to forage over larger areas than males. Interestingly, maturity stage was not a factor in the best fit model that explained space usage, indicating that space use is fairly consistent with ontogenetic shifts. In contrast, ontogeny did play a role in residency patterns, being much higher on average for juveniles, which confirms patterns seen in other rays (Hoisington and Lowe 2005). I also saw reduced space use on average in months with warmer temperatures which I speculate may result from seasonal changes in diet as seen in the closely related *T. dumerilii* (Marshall et al., 2007).

There was considerable consistency in the level of residency across sites, despite them having considerable differences in seascape characteristics. Hare Bay is characterised by expansive seagrass and soft sediment areas, Hyams is dominated by soft sediment and Bendalong is an

exposed site on the open coast. Although Bendalong is not a protected site (no spatial closure), it indicates residency to areas on the open coast and that movements are occurring on a scale relevant to existing spatial management. The consistently high residency to all these locations suggests that this is a common trait for this species in nearshore environments. Keller et al. (2017) also found relatively high short-term residency to the area surrounding an artificial reef site off Sydney, although their long-term residency to this site might have been influenced by the fact it was artificial reef habitat. This species has been shown to prefer soft-sediment habitats, but is occasionally seen on reefs (Rees, 2017).

In the context of my study, it is the proportion of individuals that exhibit behaviours that result in them being protected that is of the most interest. I show that a large proportion of individuals remain protected for extended periods. For those individuals that were not detected on the gate receivers leaving Jervis Bay ($n = 5$ left Jervis Bay) the estimates of residency and protection are conservative given that the VPS covered less than half of the soft sediment habitat in that zone. The small proportion of individuals that leave the tagging site demonstrate the potential for dispersal from no-take zones, with marine protected areas having been predicted to influence surrounding biodiversity due to the ‘spill over’ of individuals across borders (Roberts et al., 2001; Botsford et al., 2003; Gell and Roberts, 2003; Russ and Alcala, 2011). Future studies should focus on the potential for recruitment within no-take areas to influence the abundance of elasmobranchs in surrounding areas.

In summary, *T. fasciata* showed substantial residency to soft-sediment areas and the area of space use was relatively small. Resident animals repeatedly used the same areas for extended periods. The movement patterns of *T. fasciata* varied among individuals, with not all rays exhibiting the same behaviours but the majority displayed behaviour conducive to protection by spatial closures to fishing. I demonstrate the importance of collecting data on the spatial and temporal dynamics of species when designing and testing the effectiveness MPAs. These findings have implications for the conservation of batoid species and the design of marine protected areas; suggesting that no-take areas can be effective in terms of size and placement for resident batoid species. This work confirms that resident batoid species can be protected by MPAs situated in nearshore environments and challenges the assumption that soft-sediment species are all wide-ranging. Future studies should focus on determining the residency and space use of a variety of elasmobranch species in a range of environments to determine the appropriate size and position of no-take zones for the conservation and management of these species groups. Ideally movement

data would be paired with ongoing monitoring of size and abundance of taxa to determine the overall effectiveness of MPAs as a management and conservation strategy for elasmobranchs.

3.5 References

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Chapter 4 – Pregnant and Protected: no-take marine reserves can buffer the impact of capture-induced abortion in discarded elasmobranchs

Significance Statement

Fishing can cause pregnant sharks and rays to abort their pups. The effect of these abortions on population size and structure due to impacts on their recruitment is unresolved. We present a suite of evidence showing this loss of pups can have a substantial and previously unquantified impact on shark and ray populations. Our data show increased recruitment and higher pregnancy rates within no-take marine reserves for a commonly caught ray species. We also show no-take marine reserves to offer a means to quantify and potentially buffer this impact by providing areas without fishing; which can protect females while they are pregnant. These findings need to be considered in shark and ray management and conservation plans and provide support for no-take marine reserves.

4.0 Abstract

Recent research suggests that capture-induced abortion is a common, and potentially pervasive response of many elasmobranchs to fishing. The impact of these abortions on recruitment, and consequently population size and structure, remains unresolved. It appears that elasmobranchs are particularly sensitive to capture induced stress. Due to high maternal investment and long gestation periods this source of mortality may mean that the potential impacts are substantial at a population level. No-take marine reserves provide an opportunity to assess the impact of this inter-generational mortality by offering reference locations where fishing is prohibited. Assessing movements is a precursor for interpreting species response to protection. We used a 20 receiver array within a no-take area to determine long-term residency of a commonly discarded elasmobranch (*Trygonorrhina fasciata*). To determine whether capture-induced parturition affects recruitment we measured the size, abundance and pregnancy rates of *T. fasciata* in areas with and without fishing. Individuals were highly resident to small areas. Over 5 years (sampled bi-annually) juvenile abundance doubled in no-take areas while adult and sub-adult numbers remained stable, consistent with an increase in recruitment. Using ultrasonography we found a higher pregnancy rate in no-take areas which indicates that capture-induced abortions are occurring in fished areas. We provide the first indication that capture-induced abortion can have substantial population level effects on elasmobranchs. These findings are relevant to the conservation and management of discarded elasmobranch species worldwide and provide strong evidence that no-take areas can help mitigate this impact.

4.1 Introduction:

Fishing impacts shark and ray populations with declines being observed due to both targeted and incidental capture (e.g. Stevens et al., 2000; Graham et al., 2001; Stobutzki et al., 2002; Oliver et al., 2015). Capture-induced abortion, characterized by the spontaneous expulsion of an embryo before it is sufficiently developed to survive (Chapter 2; Adams et al., 2018), is a

potentially pervasive consequence of fishing which may impact the reproductive potential of live-bearing elasmobranchs (Chapter 2; Adams et al., 2018). Sharks and rays are particularly susceptible to disruptions to reproduction, having among the highest levels of maternal investment and longest gestation periods in the animal kingdom (Cortés, 2000; Dulvy et al., 2014). This response to capture has been frequently reported but the significance of these events has failed to be recognized (Chapter 2; Adams et al., 2018; Wosnick et al., 2018a; Wosnick et al., 2018b) and generally, the effects have been dismissed as minimal or insignificant. We propose that abortion due to capture represents a potential disruption to recruitment for a substantial proportion of live-bearing sharks and rays, with the impacts and magnitude of this yet to be quantified and managed.

Capture-induced abortion is of immediate concern for those species of shark or ray that are caught and subsequently discarded. These discards represent a considerable proportion of global catch (Worm et al., 2003). Discard rates are also high among recreational fishers (often termed ‘catch and release’ in this context). For example, sharks are among the top 10 key recreationally fished species in the US with > 95% being discarded (FEUS 2015). In Australia, the majority of states and territories have shark and ray catch and release rates over 95% (Woodhams and Harte, 2018). Discarding has been shown to have a suite of potential impacts ranging from post-release mortality to reduced reproductive potential through capture-induced abortion (Chapter 2; Adams et al., 2018) or reduced fitness of neonates (Guida et al., 2017; Chapter 2; Adams et al., 2018). Post-capture mortality of animals following capture and release is also a consideration that varies among species (Musyl et al., 2011). Quantifying and understanding the full impact of fishing on elasmobranch species is a requirement for designing adequate management plans for elasmobranchs (Worm et al., 2013). Considering that capture and discarding is the cause of inter-generational mortality, the only means of mitigation is likely to be adequately sized no-take marine reserves which may prevent females interacting with fishing gear; protecting them while they are pregnant (Chapter 2; Adams et al., 2018).

No-take marine reserves are areas in which all harvesting and destructive activities are prohibited and represent a common tool in conserving or restoring marine biodiversity (Gaines et al., 2010). Marine reserves are often established with conservation and/or fisheries management aims. At the same time such areas allow scientists to understand the dynamics of intact marine systems in the absence of impacts such as fishing (Gaines et al., 2010). Fish species (and other taxa) are known to show increases in both abundance and size in no-take areas (Lester et al., 2009). As a consequence, no-take marine reserves are increasingly being established globally, although most are quite small in area (Gaines et al., 2010).

Marine protected areas (White et al., 2017) and shark sanctuaries (Ward-Paige, 2017) are promoted as a means of ensuring the conservation and recovery of shark populations (Speed et al., 2018). To expect elasmobranchs numbers to respond to protection in such areas, animal movements need to be taken into consideration. Small-ranging species may be more effectively protected, whereas wide-ranging species may be protected to a lesser extent (Kramer and Chapman, 1999). Defining movement patterns is therefore a key component in ensuring no-take marine reserves adequately protect elasmobranchs (Chapman et al., 2005; Block et al., 2011; Papastamatiou and Lowe, 2012). Although many shark species are wide ranging (Heupel et al., 2005), there is growing evidence of site attachment and residency behaviour in various species (e.g. Walker et al., 1997; Hunter et al., 2006; Dewar et al., 2008; Wearmouth and Sims, 2009; Bond et al., 2012; White et al., 2017). Marine reserves therefore provide a suitable system to study the impact of capture-induced abortion and the potential for this to impact elasmobranch populations through loss of recruitment.

We know that elasmobranchs are discarded frequently, and that this can cause pregnant females to lose pups. Whether this intergenerational mortality will have a population level effects is unresolved. In this context, we aim to investigate the reproductive consequences of fisheries capture in a commonly discarded elasmobranch species. We predict that if fishing pressure is having an impact on a commonly caught and often discarded elasmobranch, then this will be indicated by higher abundances in no-take areas over time (assuming the species is resident). If this change in abundance is driven by increased recruitment, we would expect to find a difference in size structure, with a higher number of recruits in no-take areas, where females can carry pregnancy full term. In turn, we expect a lower pregnancy rate in fished areas, if recruitment is being depleted as a result of capture-induced abortion. To test these hypotheses, we firstly examined the residency of eastern fiddler rays *Trygonorrhina fasciata* to a no-take area and measured the abundance and size distribution of this species across fished and no-take zones within a marine park. Finally, we hand collected females from fished and no-take areas and conducted ultrasounds to compare pregnancy rates.

4.2 Methods:

4.2.1 Study Site

Jervis Bay Marine Park (JBMP) is a network of no-take marine reserves (sanctuary zones) and habitat protection areas located on the South-East coast of Australia and is dominated by soft sediment habitat interspersed with a range of other habitat types including rocky intertidal, subtidal reefs and seagrass beds around the Bay's perimeter (Fig. 4.1). The current zoning within

JBMP was implemented in October 2002 (Lynch 2006) with five no-take sanctuary zones where all extractive harvesting activities, including all forms of fishing, are prohibited. The remaining area of JBMP has zoning that allows for recreational fishing and limited forms of commercial fishing (i.e. bait collection and beach seine netting). A section of southern Jervis Bay is covered by the Commonwealth Waters of Booderee National Park. In Booderee Commonwealth waters, recreational fishing is permitted, however spearfishing and all forms of commercial fishing are prohibited.

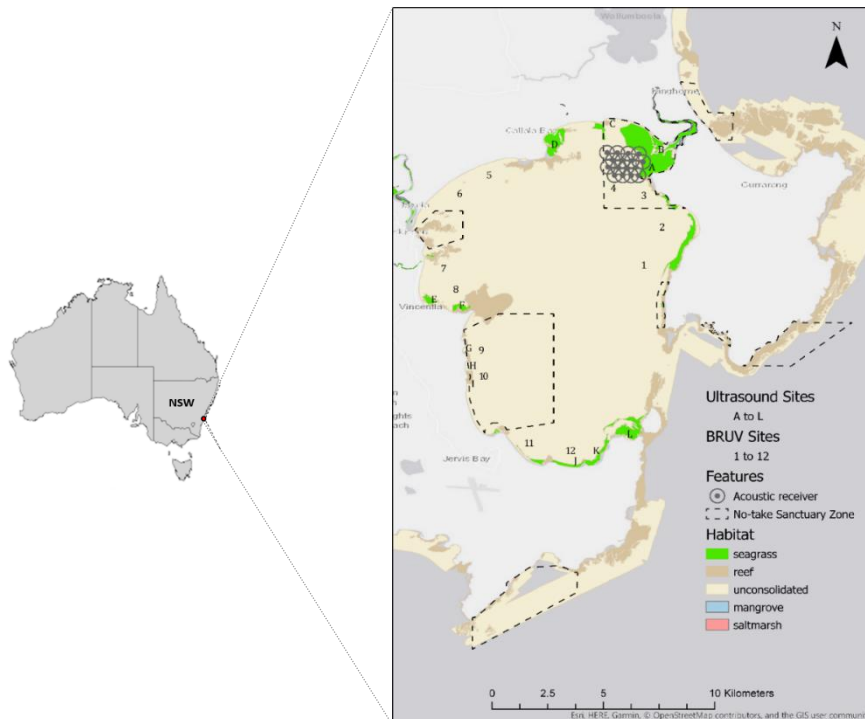


Figure 4.1 - U

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Twenty-three* eastern fiddler rays *Trygonorrhina fasciata*, were captured using rod and line with baited circle hooks within in Hare Bay no-take sanctuary zone, within range of a 20 receiver Vemco Positioning System (Fig 4.1). Ten males and 13 females ranging from 580 – 920 mm in total length were caught and tagged in three batches; in Autumn 2015 (n = 5), Summer 2015 (n = 5) and Autumn 2017 (n = 13), with surgically implanted acoustic tags using conventional methods (Barker et al., 2009). Rays were held in 50 L holding tanks and anaesthetized in seawater containing 60 mg L⁻¹ of Aqui-S. Once unresponsive, the fish were measured (TL) and a mid-ventral incision was performed after disinfection of the incision site

and surgical equipment using surgical grade iodine. A uniquely coded V9 or V13 acoustic transmitter (for tag details see Table A2.1) was inserted through a ~1 cm mid-ventral incision in the abdomen. The incision was closed with two or three dissolvable stitches tied with a double surgeon's knot. Rays were then transferred to a holding tank and monitored for recovery, before release at the site of capture. Movements were monitored between March 2015 and December 2018.

4.2.3 Abundance and size distribution

Baited remote underwater videos (BRUVs) were deployed on marine soft sediments in Jervis Bay to survey demersal elasmobranch species. Sampling was conducted in May/June of 2011, 2013 and 2015. BRUVs were deployed across all tides and were not deployed within an hour of dusk or dawn. The sampling design had four factors consisting of level of fishing (fished and no-take), year (random), and site (random), location (random), with two sites per location, and location nested within zone (Fig. 4.1). Four no-take sites and 8 fished sites were sampled with four stereo-BRUV units deployed at each site (two stereo and two single BRUV drops in 2011 and 4 stereo thereafter) giving 22 deployments in 2011, 37 in 2013 and 43 in 2015 (see Fetterplace 2017 for details). BRUVs were deployed for a minimum of 35 minutes to ensure a 30-minute analysis window. Each BRUV was deployed with 500 g of crushed pilchards (*Sardinops* sp.) with bait replaced for each deployment. A minimum distance of 200 m was kept between replicates. We measured the total length (TL) of individual rays and their relative abundance, in this case the maximum number of individuals of different lengths able to be measured in each deployment (LengthMaxN) using Event Measure 4.02. Thirty-nine fiddler rays were measured in 2011, 70 in 2013 and 108 in 2015. Individuals with a TL < 620 mm were classed as juveniles and individuals > 620 mm were classed as a combined category of adults and sub-adults. These length classes were chosen a priori based on Marshall et al. (2007) with the rationale that individuals < 620 mm TL would have claspers too small to diagnose sex from videos and is also the most conservative estimate (smallest TL) of juvenile status for our analyses.

4.2.4 Ultrasound assessments of pregnancy

In February and March 2018, female eastern fiddler rays *Trygonorrhina fasciata* were captured by freedivers at ~ 5 metres water depth from marine park zones with two levels of fishing; no-take and fished areas. The timing of this component of the study was planned to occur immediately preceding the natural timing of birth, as has been observed for the congeneric *Trygonorrhina dumerilii* (Marshall et al., 2011).

Six sites were sampled within each level of fishing and we sought to capture 10 females per site. Once captured, each individual was measured (total length and disc width) and placed

on its ventral surface in a holding tank. The scan was performed along the principal axis of the uterus using a portable veterinary ultrasound device and 3.5-MHz mechanical sector probe (V9, Beijing Carejoy Technology Co., Ltd.). To avoid using anaesthesia, *T. fasciata* were maintained in tonic immobility during the diagnosis period. An external tag was placed on captured individuals prior to release to avoid their accidental recapture. An image was taken of each ultrasound (Appendix 4), with these images assessed twice in the lab, once by the researcher who was present in the field, and again by a researcher who was not privy to level of fishing that each ray was drawn from. In this way we sought to avoid unconscious bias introduced by the knowledge of the zone each ray was taken from when diagnosing pregnancy. These two assessments were evaluated for agreement via Cohen's Kappa and McNemar's tests (Watson and Petrie, 2010). Although 103 females were ultrasounded, a storage error on the device meant images for 16 individuals were lost, reducing our verified sample size to 89.

4.2.5 Statistical analyses

Residency of fiddler rays within the Hare Bay no-take sanctuary zone was analysed using a residency index (RI). RI was defined as the number of days a tagged fish was detected, divided by the number of days monitored (Garcia et al., 2015). Days monitored ended either when the study ended or when the tag battery expired, whichever occurred first. This resulted in a monitoring period between 587 and 904 days. The rate of tagged fish loss over time was measured by plotting last day of detection for each tag against the cumulative percentage of tags remaining in the array and fitting a local polynomial regression (LOESS curve with 95% confidence intervals) (R Core Team, 2013). This was done until 603 days, determined by the last ray to leave the array (at 603 days) before the limit of 619 days which was the shortest tag deployment before the study ended.

Analyses on abundance were conducted in the software package PRIMER-E v7 with the PERMANOVA extension (Anderson, 2001). Heterogeneous variances were square root transformed to achieve homogeneity. Post-hoc pooling was done for any interaction with p-value > 0.25 to increase the power of the tests (Underwood, 1997). Dissimilarity matrices were calculated as Euclidean distances on abundances and size across samples. PERMANOVA tests were run with 9999 permutations of the residuals under a reduced model (Anderson, 2001). Size estimates were analysed following Bowman and Azzalini (1997) and Langlois et al. (2012). They were based on a null model of no difference and a permutation test. This method compares the area between Kernel Density Estimates (KDEs) from each zone to that resulting from

permutations of the data into random pairs (Langlois et al., 2012). By implementing code provided in Langlois et al. (2012) we produced plots visualising the KDE analysis. Within each plot the grey band extends one standard error above and below the mean KDE and represents the null model of no difference between the two KDEs. Pregnancy rates were analysed using binomial logistic regression (R studio, gamm4 package). We included 3 factors: Fishing level (Fished, No-take), Total Length (fixed) and Site (random). Data were tested for over dispersion prior to running models.

4.3 Results

4.2.1 Residency patterns

Residency within the 1.12 km² array was high for the majority of individuals. Thirty-five percent of the tagged rays showing a residency index of over 0.90 the > 600 day tagging period and 61% of the tagged rays showed residency over 0.50. One female (A69-1601-51355, 835 mm TL) was pregnant at the time of tagging and had a residency index of 0.997. After 365 days, more than 70% of tagged rays remained within the no-take sanctuary zone, and 50% remained after 600 days (Fig. 4.2). It should be noted that all measures of residency are a conservative estimate of protection given the receiver array covers approximately 1/3 of the area of the no-take marine reserve. Residency patterns did not appear to be related to sex or size (Fig. 4.2).

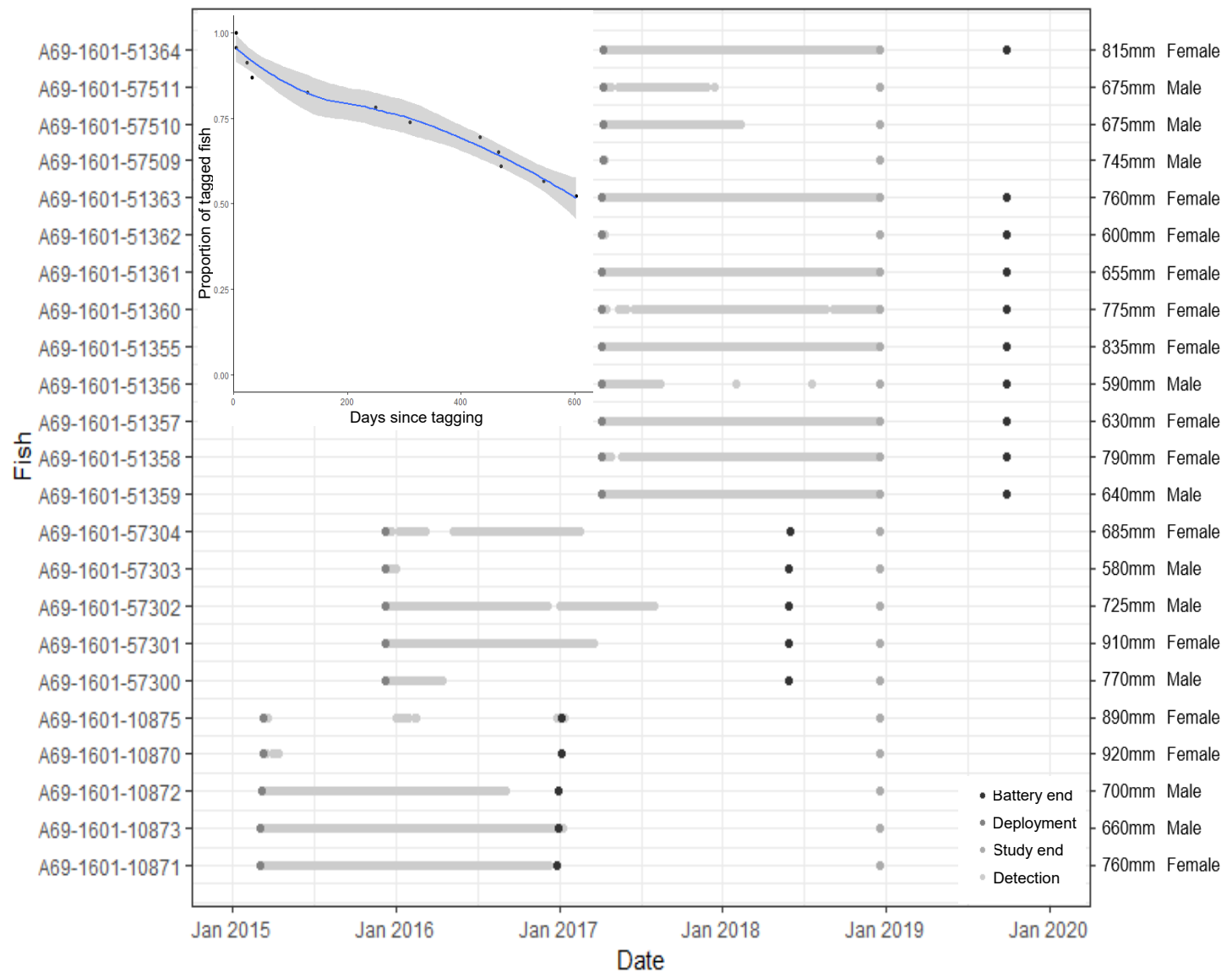


Figure 4.2- Detection plot showing the daily presence/absence of eastern fiddler rays *Trygonorrhina fasciata*, within a no-take sanctuary zone in Jervis Bay Marine Park. Data were pooled across all receivers so a daily detection on this plot could have been made on any one of 20 acoustic receivers within the sanctuary zone. No detection means the tagged ray was outside range of the 20 receivers, but not necessarily outside the sanctuary zone given the receiver array does not cover the entire no-take area. This means our estimates are conservative and provide a minimum level of residency.

The inset graph (top left) shows the proportion of tagged rays remaining in the array over time since being tagged (\pm 95% CIs). Data is shown up until 600 days at which point tag batteries began running out or the study finishes.

4.2.2 Changes in length distributions

There was a clear difference in the length distribution of fiddler rays in no-take areas in 2015 with a change in the location and shape of the length length-frequency distribution. This was driven by a shift towards smaller individuals, illustrated by the KDE function falling outside the standard

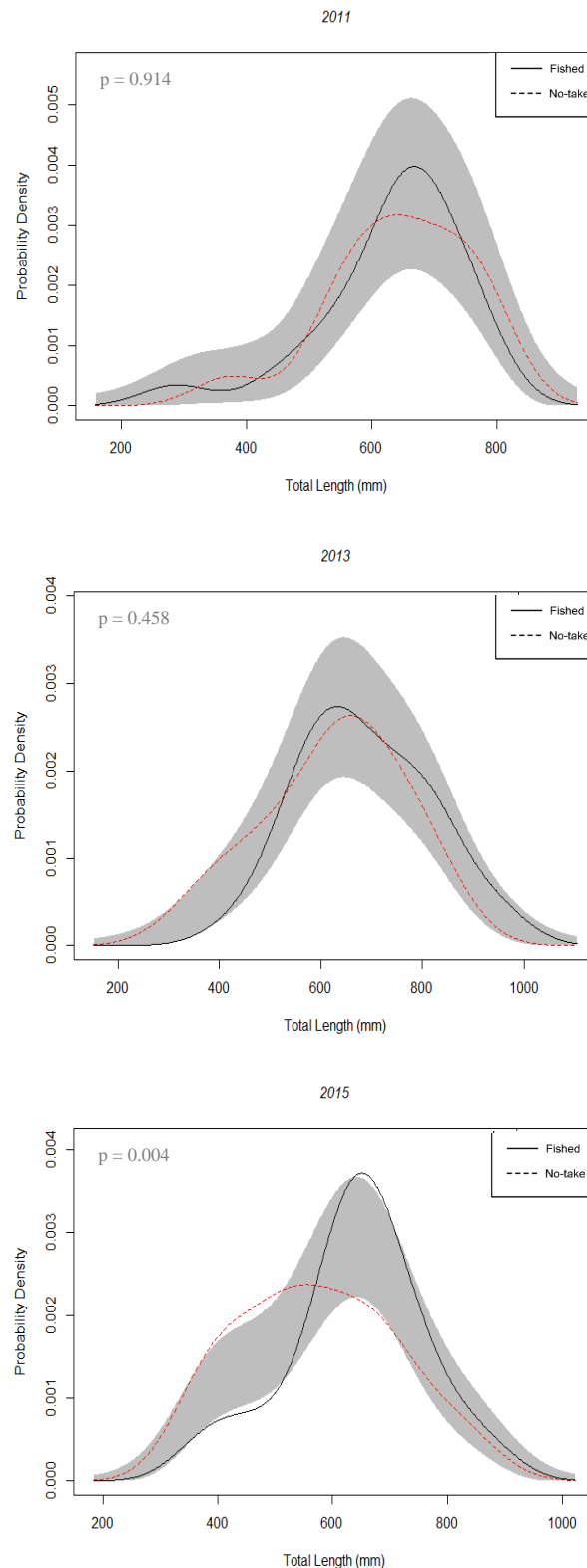


Figure 4.3 - Kernel Density Estimates (KDEs) of the length frequency distribution of fiddler rays in fished (HPZ) and no-take areas respectively for 2011, 2013 and 2015. Grey bands represent one standard error either side of the null model of no difference between the KDEs for each method.

error (the grey band which represent the null model of no difference) (Fig. 4.3). There is an apparent shift in length length-frequency through time, with 2011 having near identical length frequency distributions between zones, a slight increase in smaller individuals in SZs no take areas in 2013 (although not statistically significant) and a clear difference seen in 2015. For more detailed discussion of this shift in size distribution through time, please see section 7.4 of the general discussion.

4.2.3 Changes in abundance

Of the 217 fiddler rays measured, 94 were classified as juveniles (< 620 mm) and 123 as adults/sub-adults (> 620 mm). There was a significant increase in total abundance from 2011 to 2015 (Table A3.1) which was driven by the number of juvenile rays (Fig. 4.4, Table A3.1). Over 5 years juvenile abundance more than doubled in no-take areas from an average relative abundance of 0.85 (\pm 0.13 SE) seen per BRUV deployment in 2011 to 2.18 (\pm 0.41 SE) per deployment in 2015 (Fig. 4.4). There was no corresponding change in juvenile abundance in fished areas (Table S1). Adult and sub-adult abundance remained stable from 2011 to 2015 and across both levels of fishing (Fig. 4.4, Table A3.1).

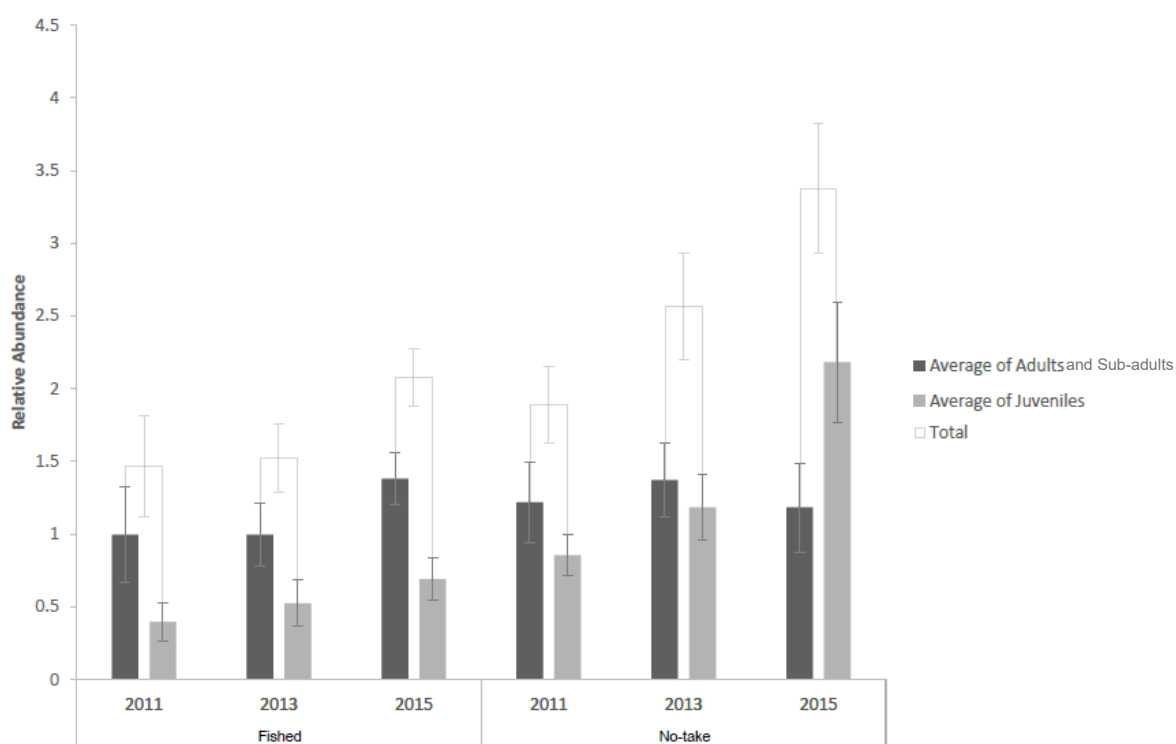


Figure 4.4 - Relative abundance of fiddler rays *Trygonorrhina fasciata* between fished (HPZ) and no-take (SZ) areas within Jervis Bay Marine Park. Data were collected via stereo BRUVs across three time periods (2011 n=22 deployments, 2013 n=37 deployments and 2015 n = 43 deployments) and abundance categories consist of juveniles (< 620 mm), adults and sub-adults combined (> 620 mm) and the sum of these two categories. Error bars are SE.

Photographic evidence of pregnancy status was available for 89 individuals. Among the females analysed, 44 were considered to be pregnant, while we deemed 45 to not be pregnant. Evaluation of pregnancy status by a second researcher showed 95% agreement between determinations (Cohen's Kappa=0.909, McNemar's chi-squared = 0, df = 1, p-value = 1). The probability of a female being pregnant was substantially higher in no-take areas (0.518 ± 0.184 CIs) when compared to fished areas (0.235 ± 0.185 CIs) (z-value= 2.113, p= 0.0346, Dispersion= 0.998) (Fig. 4.5). This pattern is consistent with our hypothesis that fishing can affect reproductive potential in elasmobranch populations. As expected, the probability of being pregnant was strongly influenced by size with larger individuals more likely to be pregnant (Fig. A3.1) (z-value= 4.105, p < 0.001, Dispersion= 0.998).

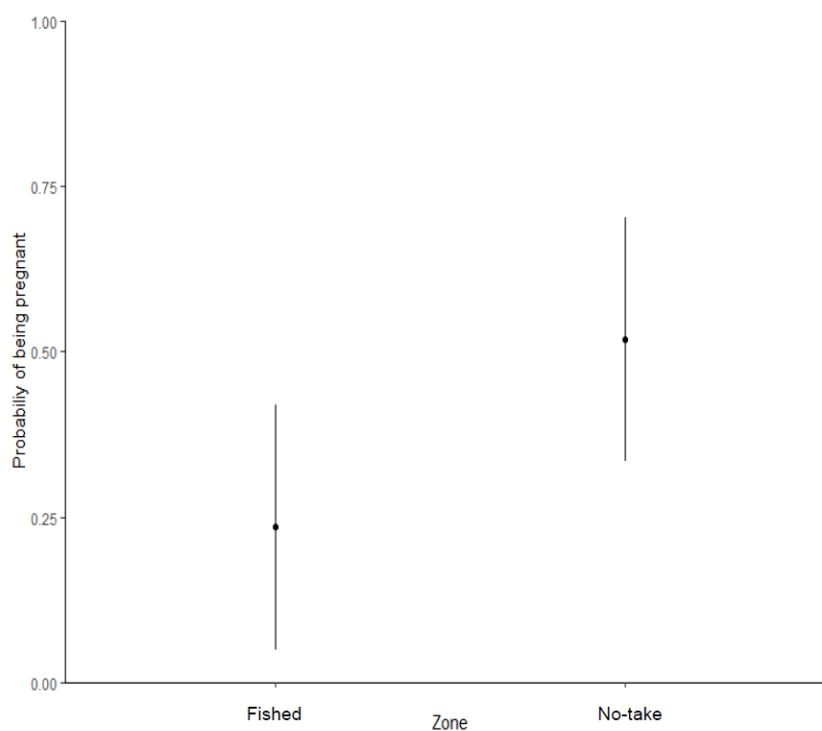


Figure 4.5 - The probability of a female fiddler ray being pregnant in fished areas and no-take areas in Jervis Bay Marine Park (n = 89 females).

4.4 Discussion

We provide the first indication that capture-induced abortion may have substantial effects on elasmobranchs at the population level. We also provide the first indication that no-take areas can potentially buffer these impacts, by providing areas where pregnant females and juveniles are protected; thereby enhancing recruitment. By using diverse methods to collect a suite of data including movement, abundance, size and pregnancy rates we provide a comprehensive and powerful test of a previously un-assessed impact of fishing in elasmobranchs. Clearly this issue is of concern for species that are predominantly discarded, which occurs frequently in recreational and commercial fisheries globally (Worm et al., 2013, FEUS 2014, Woodhams and Harte, 2018). The process of being caught can lead to a cascade of physiological reactions which have been measured in several elasmobranch species (Ellis et al., 2017). Capture stress is likely leading to disturbances to hormonal stress responsivity pathways and possibly loss of homeostatic balance; contributing to sub-lethal effects including abortion of offspring (Wosnick et al., 2018a).

We show a clear shift in the size distribution through time for a commonly caught and discarded elasmobranch; attributable to increased recruitment. It is established that at the species level, fishing can alter species abundances and size structure (Stevens et al., 2000). Notably, shifts in length composition to smaller sizes usually occur due to exploitation (Anderson, 1985; Walker and Heessen, 1996; Rago et al., 1998), yet our study demonstrates the opposite; smaller sizes in areas that are protected. This reversal is likely because our study species is heavily discarded rather than directly exploited. Larger individuals (females) are not being removed from the system as evidenced by the stable number of adults in the population. However, if they are pregnant and interact with fishing gear, they can lose their pups. In no-take areas the chance of being caught is reduced and they can carry pups full term; increasing recruitment in these areas and potentially beyond. Marine protected areas have been predicted to influence surrounding biodiversity due to the ‘spill over’ of individuals across borders (Roberts et al., 2001; Botsford et

al., 2003; Gell and Roberts, 2003; Russ and Alcala, 2011). We provide a potential mechanism for spill-over to occur via increased recruitment, but further research is required to assess if this occurs.

By examining fiddler ray movement patterns, we demonstrate that they show high residency for extended periods across both sexes and a range of sizes. Information on animal movements are of great value for management (Speed et al., 2016) with Heupel and Simpfendorfer (2005) identifying that the amount of time that species spend within protected-area boundaries is one of the most important criteria for successful reserve design. For most taxa, small home ranges and limited dispersal increases their vulnerability to impacts such as fishing (Brook et al., 2008). In turn, elasmobranchs with small range sizes are hypothesised to be more vulnerable to population decline than those that are wide ranging (Field et al., 2009) but are also the species most likely to respond to area-based protection. With *T. fasciata* spending such long periods protected, it is unsurprising that they show such a strong response to protection. There is increasing evidence that some elasmobranchs can show site attachment and residency for substantial periods (Hunter et al., 2006; Walker et al., 1997; Dewar et al., 2008; Wearmouth and Sims, 2009). For those species that may range more broadly, life stages that show high levels of site attachment are expected to experience greatest level of protection (Garla et al., 2006; Chapman et al., 2009; Heupel et al., 2010). The juvenile period is one such stage with site attachment being a precursor for juvenile elasmobranchs to show a response to a no-take area (i.e. they show limited dispersal) which we confirm in the current study. The protection of neonates and young juveniles is considered a key component for elasmobranch conservation (Kinney and Simpfendorfer 2008) and influences the design of no-take zones and marine protected areas (Heupel and Simpfendorfer 2005). Stevens (2000) highlights that net recruitment rates are important in preventing population decline, with juvenile survival appearing to be a key factor. We show no-take marine protected

areas to be a strategy for protecting elasmobranchs after birth and potentially also while they are still in the womb.

We present evidence that protection while pregnant is key to ensure un-interrupted reproduction and contribution to recruitment. Being pregnant when captured not only results in reduced recruitment but can substantially increase the chances of subsequent maternal mortality (Wosnick et al., 2018), a subject that needs immediate and targeted attention. Some elasmobranch species have gestation periods exceeding 2 years (e.g. *Squalus acanthias*; Ford, 1921) meaning they are susceptible to capture for an extended period and an abortion event represents a major loss of maternal investment. Due to their high residency, *T. fasciata* are protected while pregnant by no-take areas, leading to the observed increase in recruitment. This is evidenced by the acoustically tagged pregnant female showing very high residency for over 600 days; sufficient to cover a 12-month period of reproduction. Patterns of site attachment during pregnancy have been noted in other species with pregnant females of the critically endangered *Carcharias taurus* show strong site attachment to an aggregation site for a period matching the length of gestation (Kilpatrick and Bennet 2009). Recent studies have demonstrated that even species with expansive ranges (i.e. tiger sharks) repeatedly use areas for reproduction, thus increasing the potential of no-take marine reserves, since pregnant females are protected when they are within these grounds (Sulikowski et al., 2016).

Our comparison of pregnancy rates relies on the sanctuary zones being representative, and not placed in areas of higher diversity or selected for protection due to its attributes as a nursery. We are confident the sanctuary zones measured in this study are representative areas with comparable attributes and assemblages to the fished areas of the park. A review of the zoning plan (NSW Marine Parks Authority, 2009) found the sanctuary zones were generally appropriate for meeting the objectives of the Marine Parks Act 1997 and that those habitats and ecosystems were represented in the sanctuary zones. Fetterplace (2011) also found Sanctuary Zones to be

representative of the soft sediment fish diversity across the Jervis Bay with the fish assemblages found to be across both sanctuary zones and fished zones. We therefore confident that the patterns we are observing in pregnancy rates are most likely driven by differences in fishing pressure, rather than particular zones having attributes that attract pregnant females. Our results have implications for the conservation and management of shark/ray populations and the design of marine protected areas (MPAs). It is recognized that most existing MPAs have been established for other reasons than to protect sharks and rays, which limits their effectiveness particularly for mobile species (MacKeracher et al., 2019). Despite this, no-take marine reserves have indeed been shown to help conserve some exploited shark populations (Meekan and Cappel, 2004; Garla et al., 2006; Heupel et al., 2009; Bond et al., 2012; White et al. 2017,) and responses to protection have been seen in other discarded elasmobranch species (Fetterplace, 2017). Although large-scale MPAs may better protect wider ranging species, larger areas require increased monitoring and enforcement which can be problematic, particularly in developing countries (MacKeracher et al., 2019). Smaller MPAs or seasonal closures might be more effective during aggregation or reproductive periods (Speed et al., 2010). Pregnant females of some species are known to aggregate seasonally in shallow, warm waters (e.g. *Triakis semifasciata*, Nosal et al., 2013), making them especially vulnerable to fisheries capture, but such spatial aggregation also makes such species prime candidates for seasonal protection. Another conservation strategy, termed shark sanctuaries, have recently been implemented to curb overexploitation (Ward-Paige, 2017). While these sanctuaries are aimed at preventing direct exploitation, they cannot prevent sharks being caught incidentally and therefore cannot prevent the occurrence of capture-induced abortion.

Elasmobranchs are key species in marine ecosystems, maintaining biodiversity and ecosystem function (Bornatowski et al., 2014). Up until now, capture induced abortion has been reported for numerous elasmobranchs, but any potential population effects have been considered to be

minimal or insignificant. With little being known about the effects of discarding at the population, community, or ecosystem level (Arlinghaus et al., 2007) our findings provide indications of population level impacts and identify a clear need for further targeted research. Using a no-take marine reserve as a study system we demonstrate substantial impact of these abortions on recruitment, and consequently population size and structure. This represents an additional source of inter-generational mortality that is currently not considered in management and conservation plan. For those species already in decline, this reduced recruitment represents an additional barrier to recovery. Given the complexity bycatch management, a suite of complementary solutions will be necessary to support recovery plans. Management measures for pregnant females specifically targeting gestation, parturition and nursery areas should be considered as integral part for conservation. We hope that this study provides insights that can contribute to the continued development and assessment of tools for dynamic ocean management improve the effectiveness of shark and ray conservation.

4.5 References

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Chapter 5 – Continuous wildlife monitoring using blimps as an aerial platform: a case study observing marine megafauna

5.0 Abstract

Aerial surveys are a powerful means of collecting ecological data in terrestrial and marine systems that may otherwise be difficult to acquire. Increasingly aerial observations are made with Unmanned Aerial Vehicles (UAVs), such as drones. As this technology has improved in reliability and affordability it has replaced the traditional use of fixed-wing aircraft and helicopters. Drones do, however, have limitations, primarily in their limited flight endurance, potential to disturb wildlife and concerns over safety. Here we introduce an aerostat, a ground tethered blimp, as a logistically simple and economical alternative to drones and other aircraft. Blimps differ from drones by using helium for lift, thereby conserving battery life. This technology offers the advantage of near-continuous coverage of locations, as well as providing a safe and accessible alternative aerial platform for a range of applications. We demonstrate the viability of blimp-mounted cameras in a notoriously difficult area to conduct research; the high-energy nearshore marine zone. We identified marine megafauna, including sharks, seals, stingrays and baitfish, from the real-time video. The continuous coverage allowed the observation of foraging behaviour in sharks and seals for extended periods. Specifically, we sought to determine the likelihood of encountering marine megafauna using real-time video and whether their presence was correlated with the occurrence of baitfish. Stingrays were observed more often than other species and the occurrence of seals was correlated with the presence of baitfish. The continuous coverage allowed the observation of foraging behaviour in sharks and seals for extended periods. This demonstrates the utility of this novel technique to improve human safety and enhance ecological research.

5.1 Introduction

Aerial surveys are commonly used to sample in both terrestrial and marine ecology. Usually aerial surveys are used to obtain population estimates (Schlossberg et al., 2016; Colefax et al., 2018) but they can also be used as a tool to assess behaviour (Rieucou et al., 2018). When compared to land-based sampling methods aerial techniques have several advantages; they can provide access to remote areas (Koh and Wich 2012) or environments that are difficult or dangerous to access due to obstructions. Elevated observing platform can also improve the sightability of animals, particularly in marine environments (Torres et al., 2018, Colefax et al., 2018). Traditionally, such surveys have been conducted from fixed wing aircraft and helicopters that can be costly, noisy, and pose a risk to human safety (Tores et al., 2018). Drones, which are self-propelled (either single-rotor, multi-rotor or fixed wing) unmanned aerial vehicles (UAVs) (Domínguez-Sánchez et al., 2018) are increasingly being used as a tool to conduct and enhance ecological research (Colefax et al., 2018; Bevan et al., 2018). UAVs are proving an increasingly viable alternative to traditional aerial techniques.

Aerial video-surveillance is an emerging field with great potential and several key advantages for providing new insights into both terrestrial and marine ecology. Drones are highly mobile, easily deployable, and can be pre-programmed to collect imagery in an automated fashion. The increasing popularity and use of UAVs are likely attributable to the emergence, and increasing affordability, of a wide variety of commercial platforms which can provide high-quality real-time observations and imagery that rival or surpass data collected through traditional means (Hodgdon et al., 2018; Colefax et al., 2018). In addition, drones offer the advantage of high spatial and temporal resolution and provide a systematic and permanent record (Linchant et al., 2014). Thus, drone usage in ecology is burgeoning with a multitude of relatively affordable sensors and platforms available for scientists needing to conduct ecological surveys.

As with any tool, the use of drones in ecological research has limitations. A major limitation is their endurance in circumstances that require continuous surveillance. Smaller, affordable drones have limited flight endurance that averages 30 minutes (Hassanalian and Abdelkefi 2017), reducing capabilities for observation (Raoult et al., 2018). Further, researchers are required to have experience in operating and manoeuvring drones which can prove difficult given they must constantly monitor the field of view while keeping the drone within line of sight. Importantly, this can further reduce observational competencies by inducing observer fatigue (Raoult et al., 2018). Safety concerns for wildlife and humans also limits the use of drones in ecological research. Wedge-tailed eagles, *Aquila audax*, (and other birds) have been observed attacking drones (Lyons et al., 2017), which raises the potential for injury to wildlife and the destruction of costly sampling equipment. The safety of researchers, and the public in populous locations, is also a consideration (Fox et al., 2017) which requires detailed safety planning, training and reporting to mitigate the risk of injury. The final key constraint is one which drones share with helicopters and fixed-wing aircraft, which is the potential to disturb wildlife through the noise that they produce (Erbe et al., 2018; Mulero-Pázmány, 2017). The minimum approach distance and altitude that elicits disturbance when using a drone varies by taxa, as does the response of each taxon (Bevan et al., 2018). Due, in part, to these species-specific effects there are few scientifically justified guidelines for minimum approach distances to minimize disturbance to wildlife (Bevan et al., 2018). Although speculative, there are some indications that disturbance by drones may trigger short-term (physiological) and long-term effects (Mulero-Pázmány, 2017). Despite limitations, drones are a new platform for aerial monitoring, which offers a wide range of possibilities. Ultimately the aerial platform (and sensors) chosen by researchers depends on

study requirements, but should be fit for purpose.

To overcome some of the limitations of drones, aerostats (powered or unpowered aerial platforms using a buoyant gas for lift) may be considered more suitable aerial systems for particular applications. Aerostats have been used extensively since the early 1900s because of their long endurance and relatively low operating costs. Historical use includes military surveillance as well as intercontinental passenger transportation prior to the emergence of fixed-wing passenger aircraft (Althoff, 1990). Their use as a research tool on which to mount sensors is a more recent development. Balloons are routinely used for atmospheric monitoring (Hain and Harris, 2004) and have also been used for monitoring wildlife, including whales, dugongs and sharks (e.g. Hain, 2004; Hodgson, 2007; Nosal et al., 2012). As for other UAVs, key advantages that aerosats offer are high spatial and temporal resolution data with reduced operational costs, especially compared to planes and helicopters (Table 5.1). Some of the key advantages over drones include extended flight times and silent, non-invasive operation. Camera batteries deployed on blimps or balloons can last over eight hours, compared to the 20 to 30 minutes of a drone (Table 5.1). They are also easy to operate safely in proximity to both wildlife and humans with minimal disturbance to the animals being observed. These advantages ensure insights into patterns of movement of the target species within its habitat as well as the observation of interactions with other organisms (assuming they are visible).

This case study aims to demonstrate the capability of aerostats (in this case a blimp) as a low-cost aerial monitoring platform that can be used for near-continuous research surveillance and the assessment of human safety. Nearshore beach environments are dynamic, high-energy systems which creates accessibility and safety issues when conducting research. As a consequence, patterns of movement and the behaviour of animals in these areas is poorly understood and largely unquantified. In addition, beaches represent areas of overlap between human usage and potentially dangerous animals, most notably sharks. Therefore, a platform able to continuously monitor these zones is advantageous for public safety, as well as acquiring ecological data. In this research, we conducted an aerial survey with the purpose of quantifying the coverage achieved by a blimp-mounted camera in time and space. We also provide data on the appearance and behaviour of marine animals in the nearshore environment. Specifically, we aim to determine if certain species of marine megafauna are observed more frequently in the study area than others and whether their presence is correlated with the occurrence of baitfish. We demonstrate the application of blimps for ecological research, and the information obtained using them has

important implications for public safety as well as for scientists considering ecological survey techniques.

Table 5.1 - Comparisons of a range of aerial survey devices and their associated features. Values are ranges based on general estimates in \$AUS (note: these may vary with the specifications of the device, conditions and vary across countries). Fixed wing (based on a single engine airplane). Fuel costs are per day based on an 8 hour day

Feature	Fixed Wing	Helicopter	Rotary Drones	Aerostat
Flight time	4 to 6 hours	~ 2.5 hours	20-30 mins	8 hours
Operator requirements	Commercial pilot license	Commercial pilot license	Experienced pilot	Trained operator
Equipment cost (Aerial device)	\$75K - \$300K	\$250K – \$1.7M	\$2K - \$250K	\$5K-10K
Fuel/inflation costs	~\$1.6K day	~ \$2.4K day	NA	Relative to the length of deployment but ~\$85 day (assuming 1 week deployment)
Staffing costs	Two staff: spotter and pilot	Two staff: spotter and pilot	Typically two staff but possible with one operator	Possible with one operator
Safety	Risk of serious collision with humans and wildlife	Risk of serious collision with humans and wildlife	Risk of serious collision with humans and wildlife	Soft, small risk of rope burn during deployment and retrieval
Storage requirements	Airport or airfield	Airport or airfield	minimal	Preferably under cover (e.g. 1 car garage or shipping container)
Operational restrictions	Airport or airfield	Airport or airfield	400ft and restricted in no-fly zones	Approval required from aviation authority in operations above 400 ft and/or in no-fly zones
Mobility	Highly mobile	Highly mobile	Highly mobile	Tethered (mobile if tethered to a boat or vehicle)
Sampling method	Transect (or area based if hovering)	Transect	Transect (or area based if hovering)	Area based (or transect if towed)

5.2 Methods

This study took place in December 2017 to January 2018 in the Austral summer at Surf Beach in Kiama, on the south coast of New South Wales (NSW), Australia (Fig. 5.1). This beach is within a small (~250 m long), sandy coastal embayment enclosed by two headlands.

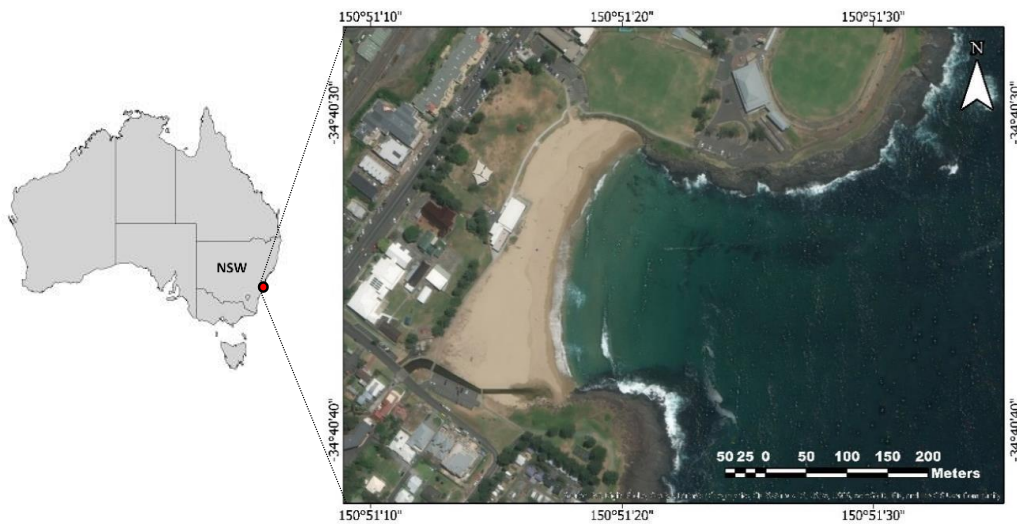


Figure 5.1- Kiama Surf Beach, a small sandy coastal embayment on the South East coast of New South Wales, Australia.

Our aim was to detect and study the behaviour and beach usage of marine fauna. We used a 5 m long and 1.8 m in diameter commercially available blimp to provide a stable platform for a high definition camera with 10 x optical zoom (Tarot Peeper) (Fig. 5.2; Fig. 5.3). The blimp was tethered at 70 m above sea-level with deployment being simple and safely achieved by the operator. Between deployments, the blimp was stored fully inflated in a garage to minimize helium usage and costs. Stored in this manner, helium loss is typically less than 1% a day so a small top-up of helium is required when the blimp lost rigidity (~twice a week). Initial inflation took 8000 L of helium which provided approximately 2 kg of lift which was adequate to lift the camera system.

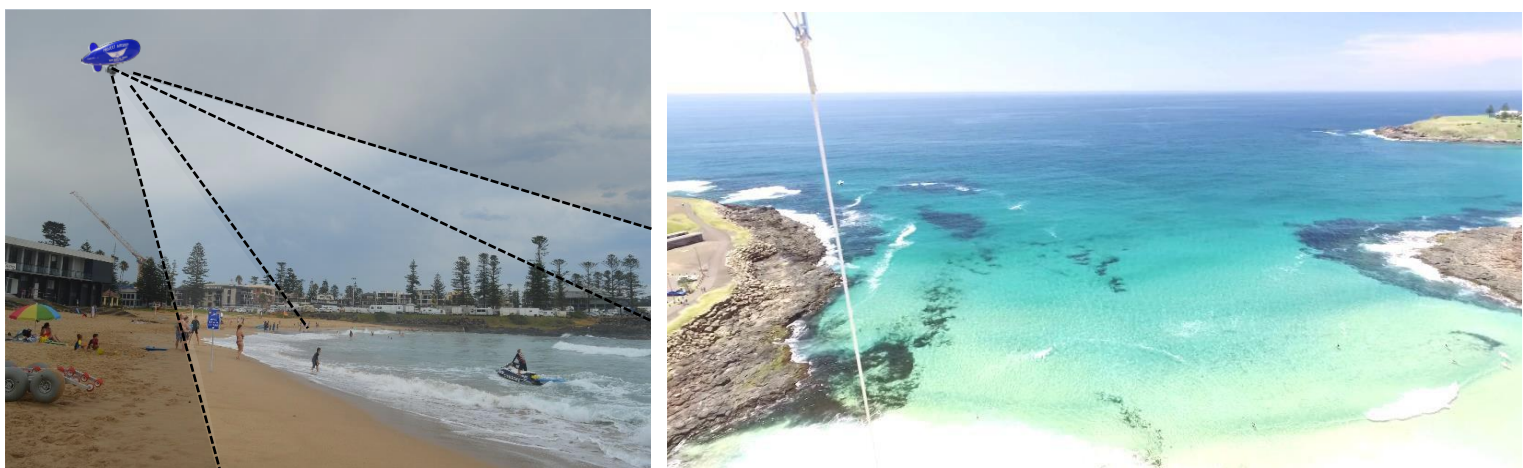


Figure 5.2 - The blimp is tethered onshore facing the water at a height of 70 m (left) to achieve camera field of view that covers the entire beach (right).

Daily surveys were conducted between 11 am to 5 pm, with some periods of sampling curtailed due to winds forecasted to exceed our 40 km/hr safety threshold. The camera sent live footage directly to a monitor on the ground using a broadcaster (DJI Lightbridge 2). The live stream was constantly monitored by an observer who controlled the camera direction and zoom (Fig. 5.3). The position in which the blimp was placed depended on the wind direction and strength. Its placement alternated between the southern, middle and northern end of the beach.

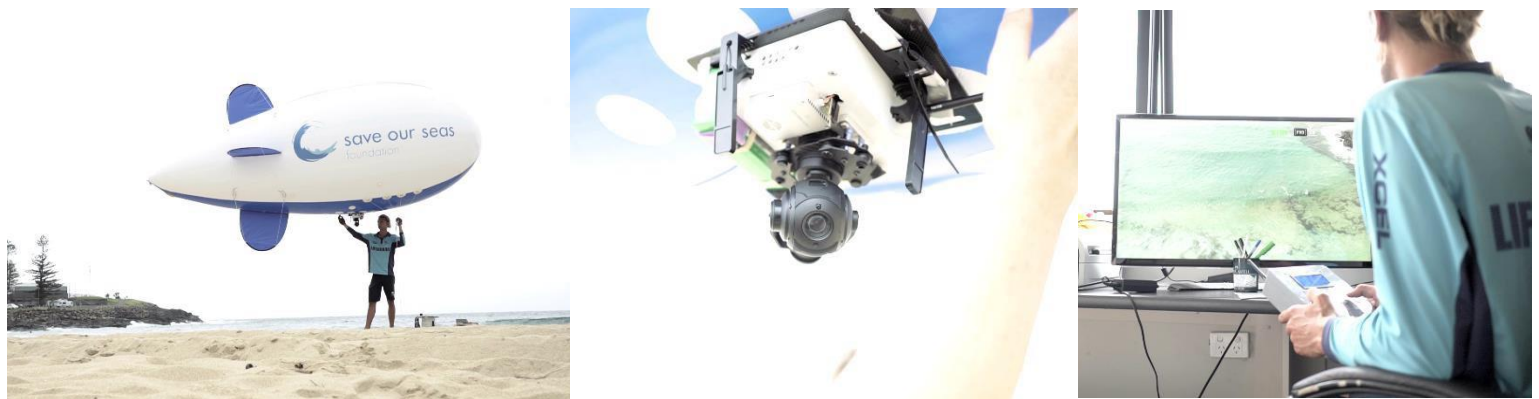


Figure 5.3 - The blimp ready for deployment (left) with camera module attached (middle) streaming to an observation monitor on the ground (right).

5.3 Results and discussion

5.3.1 Animal observation and risk prevention

The blimp was deployed on approximately 70% of days over the study period. For the other 30% of days the system could not be deployed due to high winds (> 40 km/hr) or rainfall, as the camera was not waterproof. Over a total of 16 days, we successfully observed marine megafauna from the blimp in real time. In total, 16 aerial surveys were completed with a mean daily flight time of 4 h 16 min \pm 15 min and a total of 68 h 32 min. The deployment and observation window of the blimp was determined by the work hours of lifeguards and daily flights of a shark patrol helicopter at 11 am. Grey nurse sharks *Carcharias taurus*, Australian fur seals *Arctocephalus pusillus doriferus*, and smooth stingray *Bathytoshi brevicaudata* or black stingray *Dasyatis thetidis*, were the common marine megafauna observed in our study. It was not possible to distinguish between the stingray species or among individuals of any taxon from the video recordings. The average hourly encounter rate of marine megafauna in the nearshore area of approximately 18,500 m² was dependant on the species (Fig. 5.4) and also influenced by the presence of baitfish (Fig. 5). Stingrays were observed in the study area significantly more often than both sharks and seals ($z = 5.451$, $p < 0.001$, Fig. 5.4).

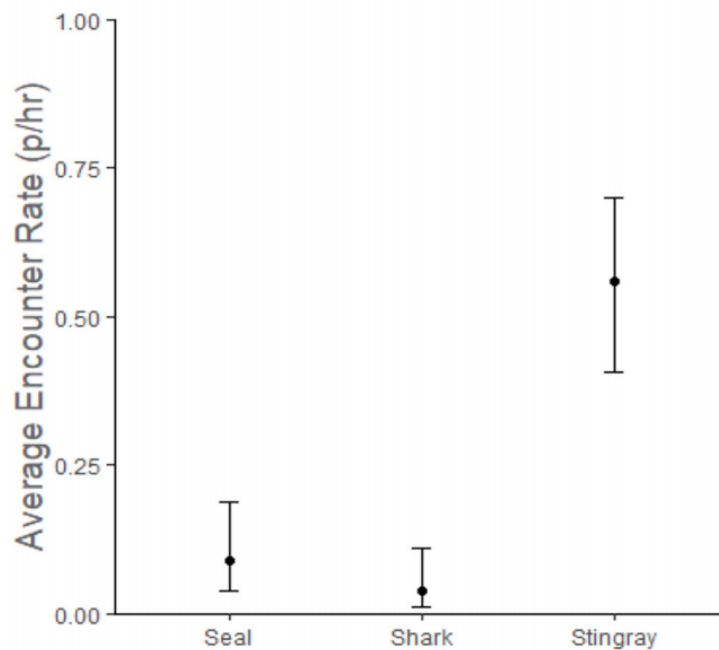


Figure 5.4 – the average hourly encounter rate in the nearshore beach environment for three mega-fauna as observed from a blimp mounted camera. The species were identified as Australian fur seals, *Arctocephalus pusillus doriferus*, Grey nurse sharks, *Carcharias taurus*, and Smooth Stingray *Bathytoshi brevicaudata* or Black Stingray *Dasyatis thetidis*. Error bars are 95 % CI.

Seals were much more likely to be encountered if baitfish were present in the bay ($z = 2.666$ $p = 0.008$, Fig. 5.5) whereas Shark ($z = 0.777$, $p = 0.437$) and Stingray ($z = 1.571$, $p = 0.116$) occurrence did not appear to be influenced by the presence of baitfish (Fig. 5.5). It seems likely then that seals use this particular beach to forage. Further, our findings suggest that the occurrence of sharks at this particular beach was very rare, and this low encounter rate may have made any correlation with baitfish activity difficult to detect. Although the seals and shark species observed here are not seen to be highly threatening, our findings have implications for humans who may seek to use these areas for recreation and adds to the recommendation by Curtis et al. (2014) to avoid entering the water when baitfish are present.

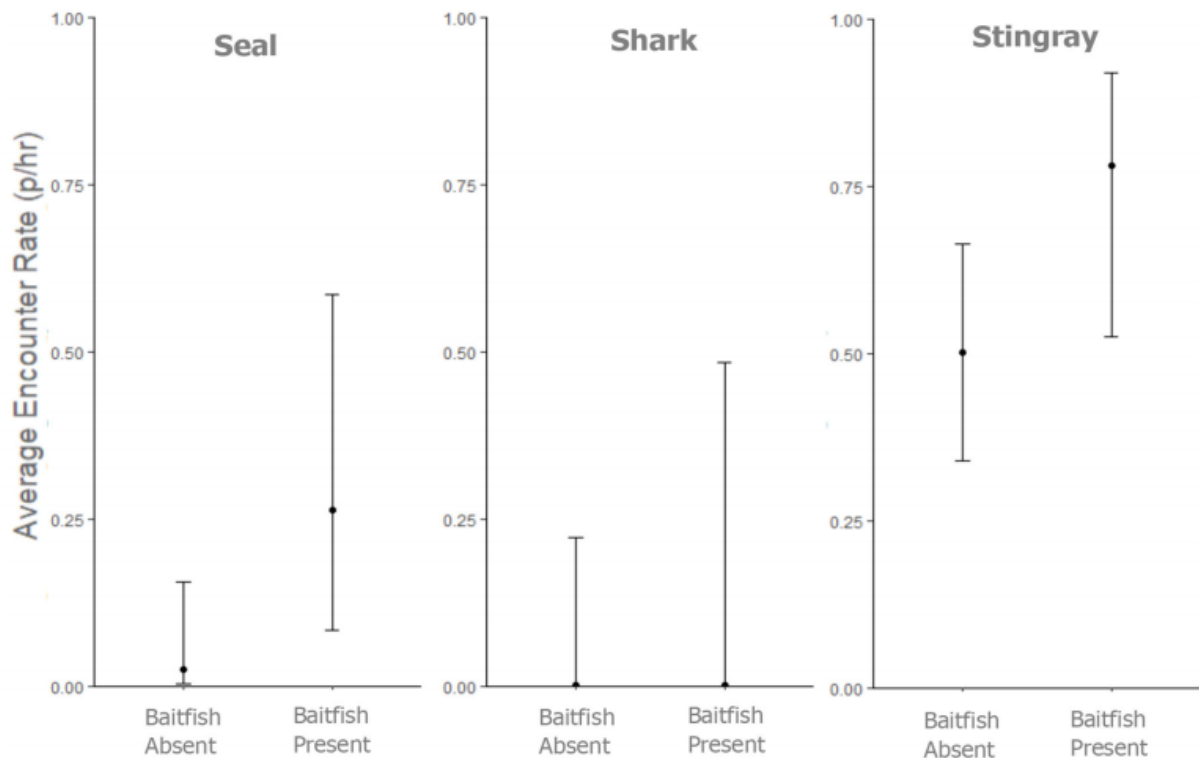


Figure 5.5 – The average hourly encounter rate of three megafaunal taxa associated with the presence of baitfish in the nearshore beach environment as observed from a blimp mounted camera. The species were identified as Australian fur seals, *Arctocephalus pusillus doriferus*, grey nurse sharks, *Carcharias taurus*, and smooth stingray *Bathytoshi brevicaudata* or black stingray *Dasyatis thetidis*. The baitfish are likely Australian salmon, *Arripis trutta*. Error bars are 95 % CI.

Unsurprisingly, our surveys observed people engaging in a variety of recreational activities including surfing, swimming, snorkelling and kayaking. One notable incident that demonstrates the application of the platform for beach safety was the observation and intervention of a bodyboarder who came close to a foraging shark (Fig. 5.6). The video observer alerted lifeguards to the proximity of the shark to the bodyboarder who were then able to signal to the bodyboarder who exited the water safely. Only later, with the acquisition of additional footage, was the species identity of the shark confirmed. Our findings highlights an important application

for aerial platforms, which could indeed be used for preventing shark-human incidents, especially given that shark incidents are known to be increasing globally (Curtis et al., 2012; McPhee, 2014).

Prior to implementation of such a platform for targeted shark detection, it would be imperative to quantify the ability of such a system to reliably detect sharks, given the limited effectiveness of other aerial shark patrols (Robbins et al., 2014).



Figure 5.6 - A bodyboarder in close proximity to a bait ball (grey mass) and associated grey nurse shark *Carcharias taurus* (top - black circle), before being alerted by lifeguards to the shark (middle- head turned towards shore) and catching the next wave in to shore (bottom). The location of the shark is indicated by the black circle.

Over the period we monitored, stingrays were observed on 98 occasions. Two large species of stingrays have been described in NSW, Smooth Stingray *Bathytoshia brevicaudata* and Black Stingray *Dasyatis thetidis*. It was not possible to distinguish between these species or among individuals from the video recordings. An unquantified number of seabirds including seagulls *Larus novaehollandiae* and cormorants *Phalacrocorax spp.* were also observed. These observations, when viewed together, paint a picture of a diverse and functional ecosystem just offshore from a populated beach. Furthermore, the insights into animal habitat usage and behaviour highlight the applicability of aerostats, including blimps, as novel tools for ecological research.

5.3.2 Behavioural observations

In addition to data on animal occurrence, continuous aerial video-surveillance provided an opportunity to collect information about the interactions of target animals with their habitat and with other organisms. One key behaviour we observed was predator-prey interactions between apex predators (sharks and seals) and their baitfish prey in the nearshore zone. On occasions, up to two grey nurse sharks *Carcharias taurus* were seen to be foraging on a baitball (Fig. 5.7). Seals were observed herding fish into the shallows and surfing waves to aid with capture of their prey. Indeed, a seal was observed to demonstrate this shallow-water herding behaviour and video recordings identified it successfully capturing a fish (Fig. 5.8). This represents the first evidence, of which we are aware, that Australian fur seals may use shallow nearshore environments as foraging grounds as Wilson et al. (2014) speculated for a different species of seal. Importantly, due to the silence and elevation of the blimp, we were able to make these behavioural observations with no impact on the animals being monitored. When using other aerial platforms such disturbances have the potential to lead to increased energy expenditure and changes in behaviour (Mulero-Pázmány 2017). This is particularly relevant and needs consideration if repeated sampling is required at one site, or sampling is focused on tracking individual animals (Raoult et al., 2018).



Figure 5.7 - Two grey nurse sharks *Carcharias taurus* attempting to feed on a baitfish school of *Arripis trutta* in shallow (approximately 2-3 m deep) water as observed from a blimp mounted camera.



Figure 5.8 - an Australian fur seal *Arctocephalus pusillus doriferus* chases a baitfish school (*Arripis trutta*) (top left), before herding them into shallow water and splitting the school into two (top right). The seal uses the shallow sandbank to it's advantage, and wounds (bottom left) and consumes (bottom right) an unfortunate salmon.

5.3.3 Shortcomings of blimp usage

As with any technique there are limitations deserving of consideration if using a blimp or balloon for research. The first relates to the costs associated with inflation of the device, as helium is quite expensive for a one-off inflation. Inflation quickly becomes economical if the blimp or balloon can be stored on site, either in a garage, shipping container or trailer. Another consideration is stability in variable winds; the blimp tends to fishtail when close to the ground where winds are more variable. We compensated for this movement by having a self-stabilising gimbal for the camera, ideally with 360-degree rotation, which automatically sustains the field of view of interest whichever direction the blimp is orientated. In marine systems Hodgson (2007), Robbins et al. (2014) and Westgate et al. (2014) have consistently demonstrated that the maximum sightable depth of large marine fauna extends 4 – 5 metres beneath the surface which, in agreement with our animal sightings, suggests this technique may be limited to surface waters or nearshore areas.

5.3.4 Prospects for continuous aerial monitoring

In the marine realm, we envision a network of such blimps, with a focus on human safety but also acting as a means for collecting continuous ecological data that would be highly valuable to researchers and coastal management more broadly. Automated detection algorithms would likely play a key role in such a network (Gonzalez et al., 2018) and could be extended to cover a variety of fauna. Further detail about animal movement patterns in this high energy environment would be ascertainable if the movement paths were georeferenced, as has been done previously (Raoult et al., 2018; Ruiz-García et al., 2018), although such analyses are beyond the scope of this particular study.

5.4 Conclusions

This study provides new insights into the behaviour that marine fauna exhibit in the high-energy surf areas and likely apply broadly to other beaches. Continuous aerial video- surveillance is a novel technique, which could provide the required information to study fine-scale movement patterns and behaviour of a variety of animals in both marine and terrestrial habitats. In terms of this case study but with relevance to other potential uses, the key advantages of using a blimp are the continuous coverage it provides, which enables the observers to detect fauna for the full day. Our platform is particularly useful in high-energy environments where other techniques may be unsuitable (Bicknell et al., 2016), or as an alternative to aerial shark spotting patrols with highly sporadic coverage and low spotting rates (Robbins. et al., 2014). In addition, the blimp operates

with zero licensing and minimal training, so it can be deployed without reference to the aviation authority and without needing a drone pilot. The costs of running such a research program can also be covered by the sale advertising space on the blimp itself. This case study adds support for the use of aerostats as an aerial monitoring platform providing insight into animal habitat usage and behaviour. Our intention has been to showcase an emerging tool for research and human safety.

5.5 References

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Chapter 6 – Coexisting with sharks: blimps provide a continuous, non-lethal alternative to traditional shark mitigation methods

6.0 Abstract

Conflict between humans and large predators is a longstanding challenge that can present negative consequences for humans and wildlife. Sharks have a global distribution and are considered to pose a potential threat to humans; concurrently many shark species are themselves threatened. Developing strategies for coexistence between humans and this keystone group is imperative. We assess blimp surveillance as a technique to simply and effectively reduce shark encounters at ocean beaches and determine the social acceptance of this technique as compared to an established mitigation strategy – shark meshing. We demonstrate the suitability of blimps for risk mitigation, with detection probabilities of shark analogues by professional lifeguards of 0.93 in ideal swimming conditions. Social surveys indicate strong social acceptance of blimps and preference for non-lethal shark mitigation. We show that continuous aerial surveillance can provide a measurable reduction in risk from sharks, improving beach safety and facilitating coexistence between people and wildlife.

“Man is not made for defeat”

— Ernest Hemingway, *The Old Man and the Sea*

6.1 Introduction

Human-wildlife conflict is a persistent and divisive issue that often results in social and environmental impacts. People can lose their sense of safety, livelihoods, and on rare occasions their lives (Thirgood et al., 2005; Nyhus, 2016). Consequently, animals can be targeted for destruction despite at times being threatened, as has been the case for the white shark *Carcharodon carcharias* (Curtis, et al., 2012) and an array of other apex-predators (Sillero-Zubiri et al., 2004). Human societies often seek to exclude predatory species perceived as threatening to human life, including lions, bears, wolves, crocodiles and sharks, despite their roles as keystone species that regulate ecological processes and maintain biodiversity (Soulé, 2013). Sharks inhabit all oceans and often frequent coastal areas that people use for recreation. The wide-ranging distribution of this group and increasing presence of humans on coastlines creates potential for conflict with humans and presents practical constraints for management.

Although the likelihood of being bitten by a shark when entering the ocean is extremely small, human perception of the overall risk is skewed by the potential horrifying consequences of a shark bite (Slovic, 1987). Shark-human interactions are also disproportionately reported in the media compared to other injuries and deaths (e.g. car accident, disease, murder), which likely

contributes to an elevated perception of danger (Gibbs and Warren, 2015; McCagh et al., 2015). Nevertheless, shark incidents are occurring more frequently (Curtis, et al., 2012; Chapman and McPhee, 2014, Mcphee, 2015; Lagabriele et al., 2018), driven mostly by rapidly growing human populations and recreational usage of the ocean. Developing strategies to support human and shark coexistence is necessary for conservation of sharks, given the dominance of anthropogenic activities in coastal areas and declines in shark populations (Stevens et al. 2000). Further complication emerges because the species of shark that bite humans tend to be species that are themselves threatened by humans (Stevens et al., 2000; Roff et al., 2018). Considering the poor condition of some shark populations globally, strategies to protect ocean users while conserving shark species are necessary to achieve sustainable socio-ecological systems in which these apex predators can exist and fulfil their ecosystem function (Berkes et al., 1998).

“Fish,” he said, “I love you and respect you very much. But I will kill you dead before this day ends.”

— Ernest Hemingway, *The Old Man and the Sea*

Globally, coastal areas are recreation and tourism hotspots. Managing shark-human interactions in these areas can be challenging, requiring management strategies that consider environmental, social and economic outcomes (Gibbs and Warren, 2015), and balance the trade-offs in selecting a given strategy. A range of strategies exist to mitigate the perceived threat to people, which vary in their impacts on sharks. Lethal strategies involve killing sharks and have been employed around the world, including in Australia since the 1930s (Green et al., 2009; Curtis et al., 2012; Gibbs and Warren, 2015) and South Africa since the 1950s (Cliff and Dudley, 2011). Culling programs have been trialled in other areas following shark incidents, including in Hawai‘i (Holland et al., 1999), Mexico and New Zealand (Neff and Yang 2013; Gibbs and Warren, 2015), but terminated shortly after starting given unacceptable environmental, social and/or economic costs. Lethal strategies most commonly include fishing methods such as gill nets (often referred to as shark netting or shark meshing in this context), and drum-lines, in cases where captured sharks of target species are destroyed. Such measures may decrease the perceived level of risk (Curtis et al., 2012), but their effectiveness has been questioned or debated (McPhee, 2012). They are environmentally ‘costly’ in terms of destruction of both target and non-target species, with bycatch to target ratios often in the order of 9:1 for gill netting in particular (Green et al., 2009;

McPhee, 2012). Due to growing public awareness of their impacts (Lemahieu et al., 2017), non-lethal methods are increasingly being proposed as sustainable and socially acceptable strategies for reducing actual and perceived risk for ocean users.

Declining public support for those traditional, lethal methods of shark mitigation has given rise to the recent popularity of modern, non-lethal technologies for shark mitigation, detection and deterrence (Simmons and Mehmet, 2018). As with traditional methods, these modern techniques are not 100% effective all of the time, and all have their limitations. Surveillance-based techniques involve continuous or intermittent observations (direct or electronic) or detections of sharks, usually with subsequent alerts and action plans after a shark is sighted (Curtis et al., 2012; Kock et al., 2012; Robbins et al., 2014; Engelbrecht et al., 2017). While these methods may provide a non-lethal management alternative to established lethal programs, they have limitations that reduce their global applicability and acceptance (Curtis et al., 2012). For example, the South African Shark Spotters program relies on spotters using binoculars positioned on high-elevation terrain adjacent to beaches to observe sharks (Kock et al., 2012; Engelbrecht et al., 2017). Aerial patrols using helicopters, although used more widely, are costly (Colefax et al., 2018), provide only short-term coverage with relatively low shark spotting rates (below 20%) and questionable effectiveness (Robbins et al., 2014). Other systems rely on a combination of shark tagging and subsequent detection on acoustic listening stations, and others on the ability of sonar arrays to determine a target shark has been observed and an alert emitted locally and broadcast via web and social media platforms (Simmons and Mehmet, 2018). Social acceptance is key to the success of any approach and there is emerging social sentiment that detection and surveillance technologies can improve efficacy of shark hazard mitigation (Simmons and Mehmet, 2018). Innovative solutions are therefore required to minimise environmental impact and maximise risk reduction. An ideal solution would provide positive coexistence and conservation outcomes, ensuring sustainable shark populations, while quantifying levels of risk and social acceptance (Carter et al., 2016).



Figure 6.1- The blimp with camera module attached (left) and the view of Surf Beach from the blimp deployed at 70 m height.

Here, we trial a novel blimp-mounted camera system and assess its effectiveness as a tool to assist lifeguards in detecting sharks at beaches (Fig. 6.1). We measure the level of risk-reduction achieved by this technique and determine the public acceptance of this novel, non-lethal approach. Our system employs relatively simple and low-cost technology harking back to the Golden Age of Flight: airships, commonly known as blimps. Historically, long flight times and relatively low operating costs resulted in the extensive use of airships for military surveillance and patrol, as well as intercontinental passenger transport (Althoff, 2016). Due to their simplicity and cost-effectiveness in providing a high vantage point and accessing the atmosphere, balloons have also been used for geographical and atmospheric research (Hain, 2000; Hodgson, 2007; Nosal et al., 2012; Chapter 5), typically carrying sensors or cameras. These lighter-than-air platforms can provide a stable vantage point for a camera with minimal power consumption; using helium for lift, as opposed to rotors, extends the battery run-time from 30 minutes (typical of UAV-mounted cameras) to over eight hours. Blimps overcome some of the short-comings of other aerial surveillance techniques including drones, which are restricted by short flight times and potential safety concerns in some populous locations (Sandbrook, 2015). Furthermore, blimps share some of the key advantages of rotary drones: they can provide data of high spatial and temporal resolution that are systematic and permanent, along with relatively low operational costs (Linchant et al., 2015). An additional advantage of using cameras for surveillance is that they provide potential for automated shark detection via algorithms (Gururatsakul et al., 2010). Blimps are also silent, easily deployed and safe in winds up to 40 km/hr with minimal training for operators. Using blimps for continuous beach surveillance could also extend the ability of lifeguards to maintain beach safety by providing an extra vantage point from which swimmers could be observed.

All techniques used for spotting fauna at sea vary in their effectiveness due to sightability errors (Colefax et al., 2018). These errors are caused by external factor biases (availability bias), and biases introduced by observers (perception bias) (Colefax et al., 2018) which need to be quantified for any survey technique, particularly one designed for public safety. Firstly, we used the performance of professional lifeguards spotting mobile shark analogues (perception bias) across different meteorological conditions and water depths (availability bias) to assess the effectiveness of our new technique. Secondly, we carried out beach-based surveys to assess the social acceptance of this new surveillance approach, compared to a lethal shark mitigation strategy. Our results demonstrate blimp surveillance to be a promising and socially accepted tool for detecting sharks in proximity to ocean users. These findings challenge previous work Robbins et al., 2014) that suggests aerial surveillance is limited in its application to shark mitigation.

“It is better to be lucky. But I would rather be exact.”

— Ernest Hemingway, *The Old Man and the Sea*

6.2 Methods

6.2.1 Study location

The two phases of this study took place at Surf Beach, Kiama on the south coast of New South Wales (NSW), Australia (Fig. 6.7) during successive austral summers: December 2016–January 2017 for trials of the blimp and camera; and December 2017–January 2018 for social surveys with the blimp deployed. Surf Beach is within a sandy coastal embayment enclosed by two headlands (~ 250 m long; Fig. 6.7). Swimmers, surfers, and fishers are frequent users of Surf Beach over summer. The bathymetry of this area continually varies due to coastal processes, but generally water depth increases progressively to approximately 1.5 m depth at 50 m offshore. At this point there is a sudden depth increase to roughly 3 m and water depth then steadily increases to a depth of ~7–10 m around 300 m offshore (Adams pers. obs., 2018). Drift-algae primarily accumulate in the north end of the bay but can be found throughout the bay. Due to rip currents adjacent to each headland, bathers are encouraged to swim only in the central zone of the bay with lifeguards providing a supervised swimming area during summer (Adams pers. obs., 2018).

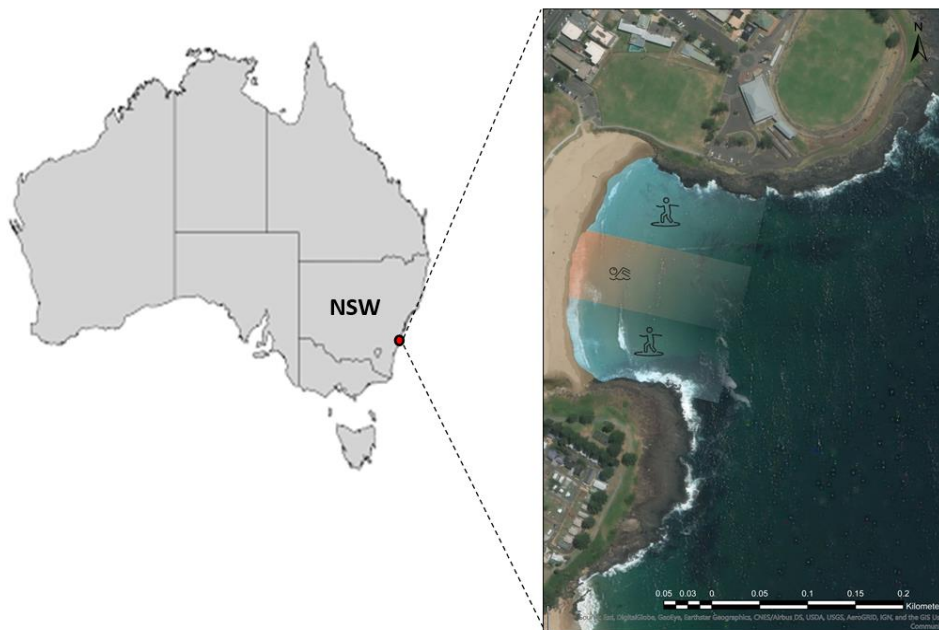


Figure 6.2 - Surf Beach, Kiama is located on the south coast of NSW Australia. The beach is typical of a sandy coastal embayment and is enclosed by two fringing rocky reef headlands. Swimmers and surfers are frequent users of the bay over summer, with usage regulated by lifeguards. Swimmers are encouraged to swim within a flagged area where surfers are prohibited.

6.2.2 Aerial platform and collection of imagery

We used a helium-filled blimp with a live streaming camera system to collect the imagery used in this study. The blimp design incorporates an 8,000 L helium-filled blimp and a gimbal-mounted camera (modified DJI Phantom 3 Advanced). Overall dimensions of the blimp were 5 m length and 1.8 m diameter with a payload of 2 kg. To minimise helium usage, the blimp was stored fully inflated with helium loss of less than 1% volume per day. The blimp was tethered above the beach at a height of 70 m, with camera settings fixed so they were consistent between trials (manual metering, ISO: 100 and Exposure: 100) (Fig. 6.1). Footage was displayed on a 24-inch monitor in the surf-club and recorded on a ground station (Atomos Ninja 2) at the transmitted 1280×720 p resolution with some image quality loss due to compression and through-air transmission. The position at which the blimp was tethered depended on the wind direction and strength, and was thus placed at either the southern, central or northern areas of the beach as necessary to ensure the blimp could observe the flagged swimming area on the beach.

6.2.3 Shark analogue deployment

To establish a shark detection rate, and how it might vary with environmental conditions, we deployed a shark analogue on 10 days across a 6-week period encompassing a variety of weather

and ocean conditions. The shark analogue was a moving free-diver of similar size to a juvenile white shark *Carcharodon carcharias* (~2.5 m including dive fins). Using a waterproof GPS watch (Garmin Fenix 3) the free-diver logged the dive starting position and swam along the bottom in a straight line parallel to the shore for 20 seconds. Although the camera had the capacity to record to an on-board memory card, all footage used in this study was recorded on the ground station to ensure any image quality loss from transmission was incorporated into the spotting rate estimate. Three shallow dives (2–3 m) and three deep dives (4–5 m) were conducted each day. Water depth was estimated by the free-divers using their body length and a dive watch when available. Environmental variables were recorded during deployment, and included cloud cover, turbidity, wind speed and wave height. Wave height was estimated subjectively by the same observer at the time of analogue deployment. Cloudy conditions were characterised by the presence of clouds covering the sun during deployment. Wind speed was estimated from a weather station deployed at the study location and converted to a two-level categorical variable for analysis. Turbidity was consistent throughout the study period with the secchi depth never shallower than the deepest analogue deployment (5 m).

6.2.4 Lifeguard shark analogue spotting trials

Footage collected during the trial was compiled and later shown to 20 professional lifeguards on a 24-inch HD monitor in a double-blind trial. Footage shown to lifeguards consisted of 104 randomised video clips (20 second duration); 51 having a shark analogue present and 53 showing the same beach with the shark analogue absent. The 20 second clips compiled with the analogue absent were haphazardly selected from the 15 minutes prior or 15 minutes after the shark analogue was deployed each day to ensure minimal change in environmental conditions. Participants were asked to determine whether the shark analogue was present or absent, and to avoid subconscious prompting, the invigilator of the trial was not privy to the correct classification.

6.2.5 Social surveys

Surveys were conducted to gauge public opinion of two measures for shark mitigation: (i) the novel use of a blimp-mounted camera; and (ii) of more traditional approaches to shark mitigation. Questionnaire surveys were conducted face-to-face at the beach while the blimp was flying, to assess real-time attitudes towards the blimp. Questions focused on four elements relevant to this study: beach use; sense of comfort and safety with the blimp; general views on shark hazard mitigation; and views on the New South Wales *Shark Meshing (Bather Protection) Program*, which has been in operation since 1937. Broad demographic data were also collected. The beach surveys were conducted over three days in January 2018, during the NSW school holidays (peak

beach usage) while the blimp was deployed. Kiama and the south coast are holiday destinations, so at this time local residents and tourists (predominantly from the state of NSW) visit the beach. Beachgoers were offered paper and electronic versions of the survey, which included a standardised briefing on shark meshing. Electronic versions were provided on tablets using the program SurveyMonkey. Beachgoers were approached directly by one of the researchers and asked if they were willing to participate in the research. Here we present basic descriptive analysis of survey responses. Survey design and procedure were approved by the University of Wollongong Human Research Ethics Committee (HREC number 2016/993).

6.2.6 Data analysis

We used two approaches to assess the performance of the blimp. First, we measured the average detection probability across lifeguards to assess external factor bias introduced by the environment, and second, we measured observer error (known as perception bias) by quantifying and comparing accuracy within and among lifeguards using signal detection theory.

6.2.7 Shark analogue detection probability (availability bias)

To create a shark detection probability map that models the level of safety achieved at a beach by blimp surveillance, we used simple kriging to create two interpolated surfaces for sunny and cloudy conditions with analyses conducted in ArcGIS Pro version 2.0. Interpolation was based on the position of the sunny ($n = 29$) and cloudy ($n = 22$) shark analogue deployments and the proportion of each deployment that was spotted by the 20 Lifeguards. That is, if 20/20 lifeguards detected an analogue deployment, this point was assigned a value of 1 and, if 0/20 detected a deployment, that point was assigned a value of 0. To satisfy the assumptions of kriging in terms of error assessment, the data were arcsine transformed prior to interpolation. To make both maps comparable, the exploratory trend surface was standardised, and exponential polynomials were used to de-trend the data prior to fitting semi-variograms. In order to interpolate the risk-reduction map to cover the entire bay, two points were seeded at the high-water mark with a spotting value of one, and two points were seeded ~300 m offshore from the beach with a spotting value of zero. The interpolated surface was then truncated using the coastline as a barrier so that predictions only occur in water.

To compare the shark analogue detection probability under different environmental conditions we used a generalized linear mixed model fit by maximum likelihood (Laplace approximation) in the logit binomial family using the lme4 package in R (R Development Core Team, 2008, Bates et al., 2012). Light conditions were included in the model as a two-level fixed factor (Sunny, Cloudy), and were crossed as a three-way interaction with water depth and wind speed,

which were also two-level fixed factors; Shallow and Deep, and ≤ 10 km/h and 11 – 20 km/h respectively. Wave height was included as a random factor with 4 levels (1 – 4 feet) as we wanted to account for any potential variation attributable to wave height in the model. Lifeguard was included in the model as another random factor to control for variation among observers. Significant interaction terms were further investigated using Tukey’s post-hoc multiple comparisons of means (Horthon et al., 2008) using the ‘multcomp’ package (Horthon et al., 2016).

6.2.8 Assessing Lifeguard Accuracy (perception bias)

To assess the accuracy of the system we assessed the detection sensitivity, bias, and false alarm rate of lifeguards between two light conditions (Sunny vs Cloudy) using signal detection theory (Stanislav and Todorov, 1999). A yes/no trial in signal detection theory results in one of four possible outcomes (Table 6.1).

Table 6.1 - Signal detection outcomes based on the presence or absence of shark analogues, visual ‘noise’ and the response of lifeguards

Shark analogue				Lifeguard response		Outcome
present						
Yes	+	noise	+	positive	=	Detection
Yes	+	noise	+	negative	=	Miss
No	+	noise	+	positive	=	False Alarm
No	+	noise	+	negative	=	Correct Rejection

Sensitivity to stimuli (d') provides a summary of the ability of lifeguards to distinguish shark analogues (signals) from visual ‘noise’ (Stanislav, and Todorov, 1999). The higher the value of d' the more sensitive a lifeguard is to stimuli (Stanislav, and Todorov, 1999). Signal detection theory metrics were calculated for each lifeguard in sunny conditions and cloudy conditions. A standard correction was applied when the hit or false-alarm rate equalled 0 or 1 (Macmillan and Kaplan, 1985; Stanislav and Todorov, 1999). Depth was unable to be included in these analyses, as no value for water depth could be assigned when the shark analogue was absent. The average values of sensitivity (d') and bias (C) were compared using paired t-tests to account for inherent differences in individual lifeguards.

6.3 Results

6.3.1 Quantifying detection probability in a variable environment

The highest probability of lifeguards detecting shark analogues were in nearshore areas where swimmers usually frequent ocean beaches (Fig. 6.2). Detection probability decreased with increasing distance from shore until it approached zero. The probability of lifeguards detecting analogues in sunny conditions (Fig. 6.2a) was generally higher than in cloudy (Fig. 6.2b), and this was particularly apparent in the nearshore region (Fig. 6.2).

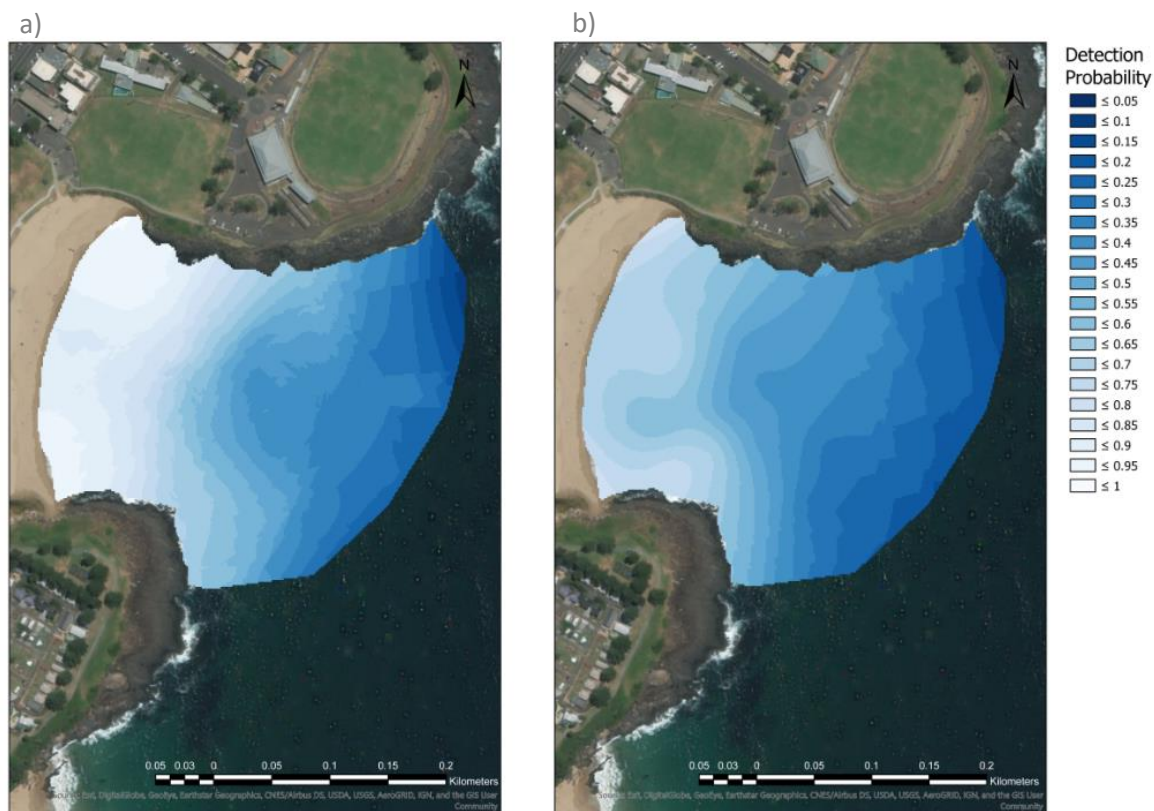


Figure 6.3 - Shark analogue detection probabilities in a sunny (n=29 analogue deployments over 5 days) and cloudy (n=22 analogue deployments over 5 days) conditions at Surf Beach in Kiama, NSW, Australia.

Understanding how the probability of detecting sharks changes with environmental conditions is necessary if surveillance is to be relied upon for human safety. Shark analogue detection probability was extremely high (0.925 ± 0.0334) in ideal swimming conditions (i.e. sunny days and low winds) across the areas where beachgoers tend to congregate (shallow water within the patrolled swimming area) and remained effective in stronger winds (Fig. 6.3). With greater water depth, analogue detection on sunny days with low winds was similar to that of shallow water with sunny days and higher winds (Fig. 6.3). Unsurprisingly, detection of analogues was reduced for both depths with increasing cloud cover and in windy conditions (Fig. 6.3). Detection probability

was lower when winds were stronger (Fig. 6.3), but swimmers tend to be far less common in such conditions. Patterns in detection probabilities in deep water deployments generally mirrored those in shallow water but were reduced to a greater extent (Fig. 6.3). Detection in deep water and cloudy conditions with high winds was lowest and should be considered unreliable (Fig. 6.3).

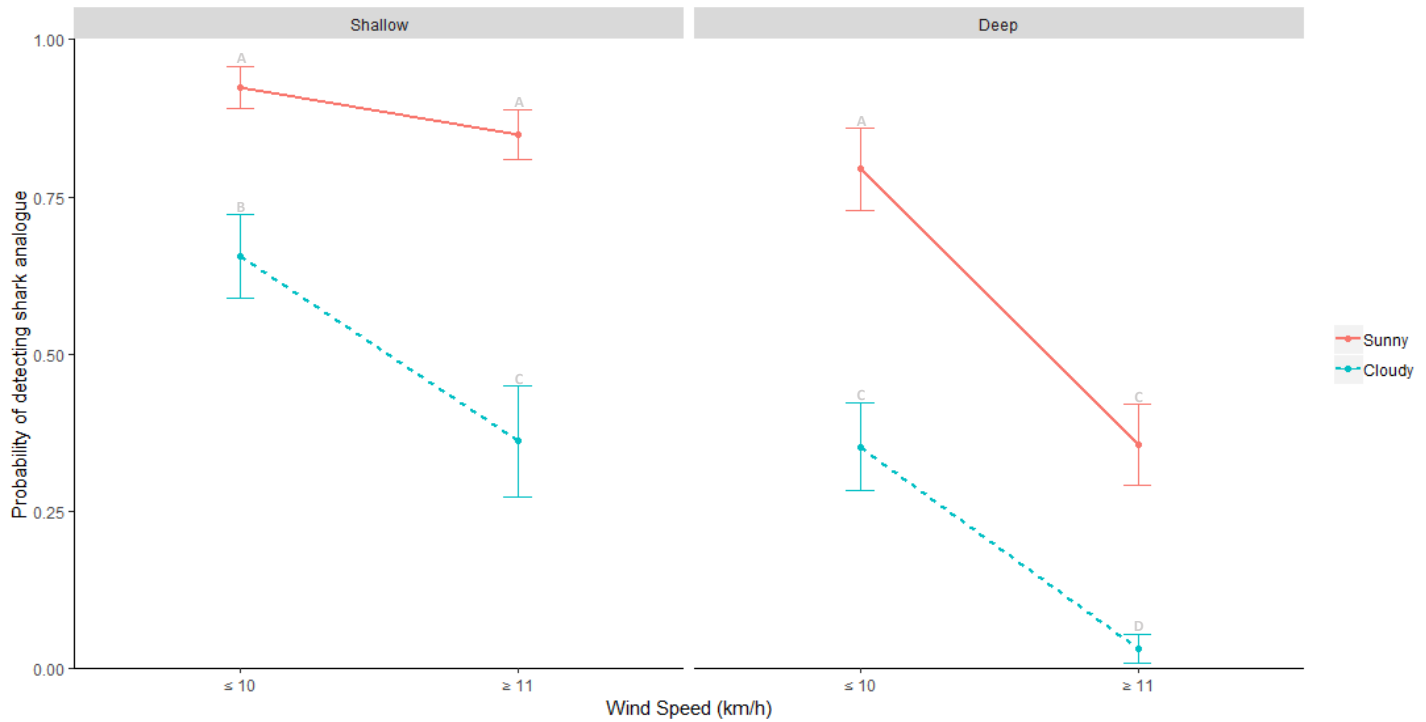


Figure 6.4 - The probability of detection of 51 shark analogue deployments determined in trials by 20 lifeguards (\pm SE) in a double blind trial based on water depth (shallow: 2 - 3 m and deep: 4 - 5 m) versus cloud cover (sunny versus cloudy) and wind speed. Letters denote significant differences as determined by post hoc Tukeys multiple comparisons of means.

6.3.2 Assessing Lifeguard Accuracy

The human element (perception bias) of shark detection is important to quantify, especially in changeable environmental conditions. Accuracy in this context is a measure of individual observer performance when using the novel technique and forms part of assessing the applicability of the whole shark-detection system. We quantified perception bias on two levels: sensitivity and false alarm rates.

A higher detection sensitivity (d') indicates easier visual discrimination of shark analogues from background 'noise' such as drift algae. The d' of lifeguards to detecting shark analogue presence was greater in sunny conditions than with cloud cover ($t = -2.83$, $df = 19$, $P = 0.01$) (Fig 6.4a). False alarms represent occasions when a lifeguard indicated they had seen a shark analogue when none was present, most likely resulting from observers spotting drift algae. False alarm rates were

unaffected by cloud cover ($t = -1.71$, $df = 19$, $P = 0.10$) and quite variable among lifeguards (Fig 6.4b).

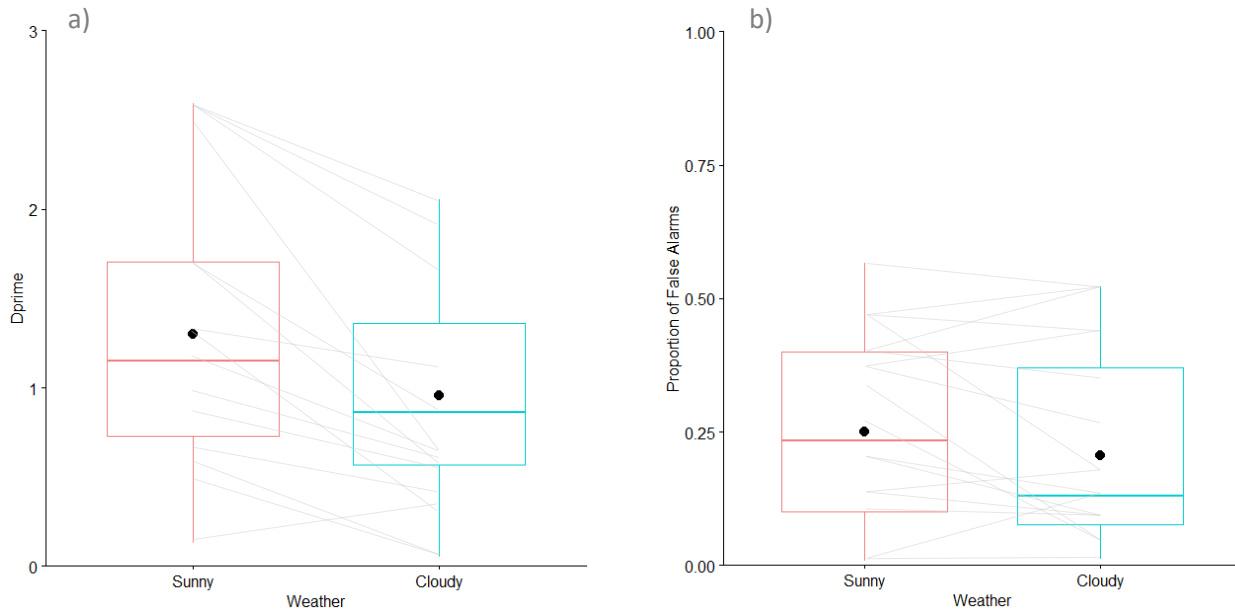


Figure 6.5 - (a) shark analogue detection sensitivity and (b) the proportion of false alarms of 20 professional lifeguards in Sunny and Cloudy conditions. Black point indicates the group mean. Grey lines join each individual lifeguard across the different weather conditions. A higher d' indicates the easier discrimination of shark analogues from background noise (e.g. drift algae).

6.3.3 Social acceptance

Results of our survey completed by 115 beachgoers suggest wide acceptance of aerial surveillance by blimps. Beach activities varied among respondents; the majority reported usually using the beach for recreation, including swimming, body-boarding, body-surfing or playing in the breakers (71%). Seventy-four per cent of respondents had entered the water on the day they completed the survey. To determine the level of public acceptance of aerial surveillance using a blimp, we investigated two measures: sense of safety; and degree of comfort. On a five-point Likert scale, 84% of respondents said the blimp made them feel much safer (45%) or a little safer (39%); 16% reported feeling no different; none reported feeling less safe (Fig. 6.5a). Ninety percent of respondents reported feeling very comfortable (80%) or fairly comfortable (10%) with the blimp conducting aerial surveillance at the beach; 9% were neutral; and 1% (one respondent) reported feeling very uncomfortable (Fig. 6.5b). Sixty-seven percent of respondents answered they would ‘choose to go to a beach with a blimp rather than one without, if both beaches were good and convenient’; 20% were undecided; and 13% reported they would not.

Ninety percent stated they would like to see blimps at other beaches to improve beach safety; 10% were undecided.

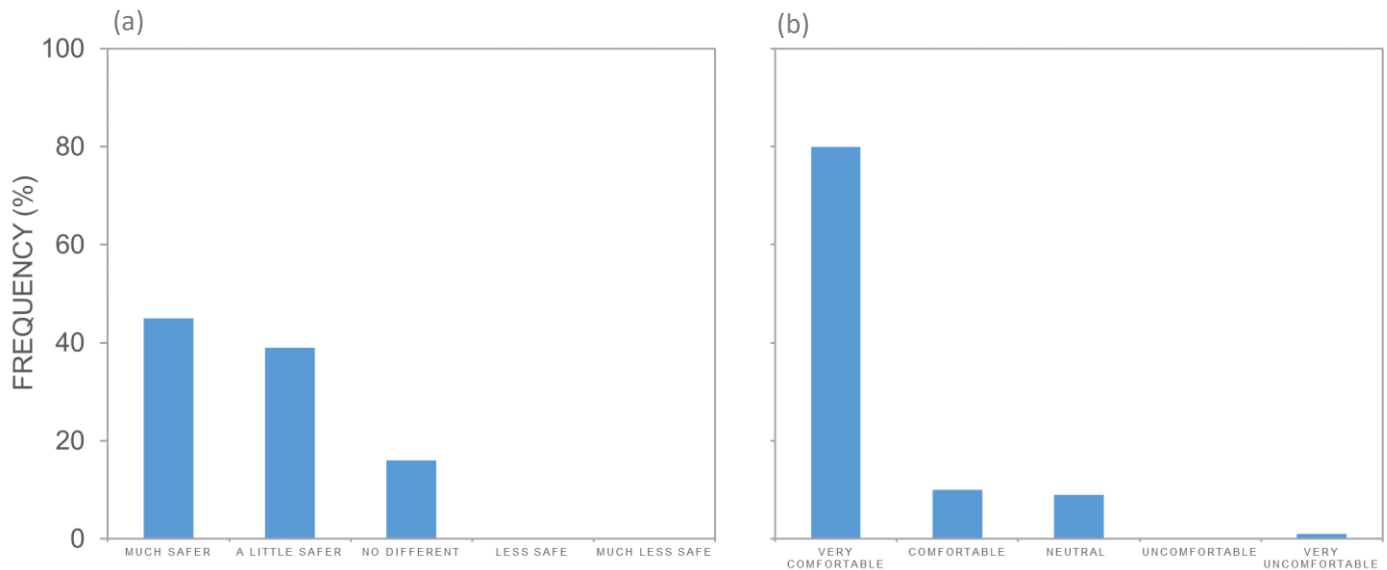


Figure 6.6 - Frequency histogram showing responses to two related survey questions: **(a)** 'Does the blimp give you a sense of safety from beach hazards?' and **(b)** 'Do you feel comfortable with the blimp at the beach?'

These levels of social acceptance compare favourably to shark meshing (the longest running form of shark hazard management for bather safety) (Green et al., 2009). Respondents had mixed feelings regarding shark meshing. On a five-point Likert scale most respondents (39%) neither supported nor opposed, or had mixed feelings about shark meshing; 33% opposed (22%) or strongly opposed (11%); and 27% supported (18%) or strongly supported (9%) shark meshing. The sense of safety offered by shark meshing was mixed: 41% reported that it did not make them feel safer; 32% reported it did; 26% were undecided. It is important to note that the study site does not have shark meshing in place, and respondents may not have detailed knowledge of what shark meshing entails. Forty-five percent of people did not want to see shark meshing extended to cover the study area; 28% did and 27% were undecided. Respondents were asked what approaches they would like to see for keeping people safe from potential threats from sharks, from a list of five possible commonly adopted approaches; multiple approaches could be selected. The vast majority (93%) selected 'Spotting or detecting sharks, through methods that do not harm them'. Least popular strategies were 'Catching and killing sharks' (2%) and 'Catching sharks and taking them off-shore, even if there is a risk of harming them' (10%) (Fig. 6.7).

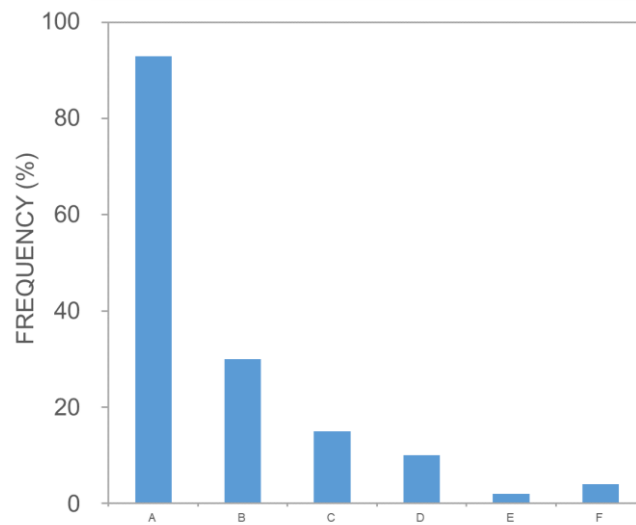


Figure 6.7 - Frequency histogram showing responses to the survey question: ‘In general, what approaches would you like to see for keeping people safe from potential threats from sharks? (Choose as many options as you like)’ where A is ‘Spotting or detecting sharks, through methods that do not harm them’; B is ‘Relying on individuals taking responsibility for their own actions’; C is ‘Relying on personal deterrent devices, like electrical shields’; D is ‘Catching sharks and taking them off-shore, even if there is a risk of harming them’; E is ‘Catching and killing sharks’; and F is ‘Other’.

6.4 Discussion

“he knew no man was ever alone on the sea.”

— Ernest Hemingway, *The Old Man and the Sea*

Blimps can provide accurate, continuous and cost-effective aerial detection of sharks at beaches, offering lifeguards a unique and powerful tool for improving safety. Detection of shark analogues was most effective in shallow water (< 2m)—the areas most commonly frequented by swimmers (Adams, pers. obs. 2018¹), and in sunny conditions—the time that beach visitation is highest (Kammler and Schernewski, 2004; Moreno et al., 2008). Higher wind speeds generally meant lower detection probability, which we attribute to two factors: increased camera movement and surface chop. When viewed collectively, this study highlights the practical application and limitations of this new technology for providing a measurable level of shark detection at ocean beaches.

¹ Adams has worked as a Professional Ocean Lifeguard for 7 years.

Accuracy is key to the success of any novel technique to reduce potential interactions between sharks and bathers. Accuracy assessment has relevance to managers seeking to implement new technology, as well as observers themselves seeking to meet minimum performance standards. The high sensitivity of lifeguards to analogues deployed on sunny days provides further support for blimp-based aerial surveillance during popular beach use conditions. False alarms were most likely attributable to mobile drift algae, frequently observed to move slowly through the study area. In reality, a glimpse of this type would attract further scrutiny to confirm the observation. These trials were based on discrete events (20 seconds), while in a real world scenario an object suspected of being a shark would be further observed to confirm identification. A shark is likely to be observed at several points in time as it moves through the area covered by the field of view of the blimp (250 m²). Typical cruising speeds of large coastal sharks are below 1 metre per second (Ryan et al., 2015) providing observers' sufficient time to view a shark in the "buffer zone" prior to it coming into direct contact with humans.

Blimp-mounted cameras can outperform established methods of aerial surveillance, such as planes and helicopters, in several key areas. Importantly, the coverage and shark detection capacity they provide is continuous rather than restricted to the few minutes per beach per day for helicopter and fixed wing patrols. Unlike planes and helicopters, the coverage of a blimp is area based at 250 m², therefore complete coverage of longer beaches could not be achieved, rather beach users could be encouraged to utilise the area that the blimp is located (typically over designated swimming areas or a surf break). In terms of risk-reduction, the probability of detecting sharks from the blimp was high (93%) in ideal conditions. This can be compared to results of fixed-wing aircraft and helicopter assessments, which achieved shark analogue spotting rates of 13% and 17% respectively with analogues deployed at comparable depth below the surface (Westgate et al., 2014). The generally higher detection rates from the blimp are likely due to several factors, including that the blimp is stationary, while the aircraft travel at over 30 m/s (110 km/h) making spotting a small target in the water difficult (Westgate et al., 2014). The shark analogues in Robbins et al. (2014) were stationary, while our analogues were moving, providing a more representative model of shark behaviour. The blimp was targeting shallow (2–5 m) near-shore swimming areas, which may increase the silhouetting of the analogues compared to the fixed-wing and helicopter assessments carried out by Robbins et al.; their analogues were deployed at ~2 m below the surface in water 6–12 m deep. Detection rates of our shark analogue dropped in cloudy conditions and in deeper deployments. Robbins et al. (2014) and Westgate et al. (2014) also reported reduced detection ability at depth and showed the maximum deployment

depth at which an analogue was detected to be 4.3 m and 5 m when assessed by fixed-wing aircraft and helicopter respectively.

Shark mitigation demands balanced consideration of beach safety, species conservation (Pepin-Neff, 2016), and social acceptance. We found extremely high public acceptance of blimps for beach safety, and strong preference for non-lethal strategies. Acceptance comprised two elements: the blimp provided a sense of safety and respondents felt very comfortable in its presence. The majority of respondents stated they would choose a beach with a blimp for aerial surveillance if all other factors were equal, and a large majority reported a desire to see blimps at more beaches to improve safety. In comparison, reported support for shark meshing was mixed; with lower support than observed for blimps. Few people reported that shark meshing made them feel safe, which was surprising given safety is the aim of the program. Respondents reported very strong preference for methods that detect sharks without harming them, suggesting people are accepting of non-lethal approaches; a finding consistent with other studies (Gibbs and Warren, 2015; Pepin-Neff, 2016; Simmons and Mehemet, 2018). Indeed, in a shark mitigation strategy preference analysis, Simmons and Mehemet (2014) reported that the likelihood of harm to sharks and other marine species is a central reason for community preferences. Importantly, respondents in our study were casual beachgoers who use the near-shore area for recreation. We believe this group represents the majority of beach-users and gives our findings strong relevance to managers and policy makers. Several studies highlight the dilemma faced by managers tasked with mitigating the likelihood of shark incidents, and the reliance on established strategies that offer perception, but limited evidence, of risk reduction (Crossley et al., 2014; Gibbs and Warren, 2015; Gray and Gray, 2017; Lagabriele et al., 2018).

Our results indicate the potential for a substantial reduction in the likelihood of a shark interaction for activities in the surf zone covered by the blimp, assuming bathers exit the water if a shark is sighted. Our technique is most applicable to activities held in a specific location (e.g. flagged swimming areas, surf contests, surf carnivals etc.). There will always be areas that cannot be patrolled due to remoteness and insufficient resourcing. Personal shark deterrents may offer an alternative strategy to reduce the likelihood of shark incidents for ocean-users outside patrolled hours or in remote locations (Huveneers et al., 2018). The blimp also has the proven ability to tap into current and emerging community-based shark alert systems that use acoustic telemetry or surveillance linked to smart-apps to warn of shark presence at beaches (Anonymous 2018 a,b,c,d,e). Blimp-based aerial surveillance shows promise as a highly visible, easily communicated and socially accepted shark hazard mitigation strategy.

Further development could be made with the use of multispectral cameras with blue light filters, which have been used previously to monitor sub-surface whales from space (Fretwell et al., 2014). There is little doubt that continuing improvements in this area could be made using emerging remote sensing technologies, which are evolving rapidly. Research is required regarding the impacts of water clarity on shark detection rates, and the impact that time of day may have on light levels and consequent spotting ability. The effect of time of day on spotting rates is particularly important to define, given the possibility of increased risk in shark incidents during dawn and dusk (Curtis et al., 2012). It is important to note that beach visitation has been shown to peak during the middle of the day (Kammler and Schernewski, 2004) and beaches are unpatrolled at dawn and dusk (Adams pers., obs. 2018). The results of our study demonstrate that blimp surveillance has direct and immediate application to global shark hazard mitigation and offers managers both a proactive strategy and a means of rapid response following shark incidents.

We conclude that blimps as an aerial surveillance technique, are demonstrably effective and represent a simple, cost-effective and socially accepted tool for mitigating the risk of shark incidents at ocean beaches. Lethal strategies have obvious environmental risks and ethical dilemmas (Gray and Gray, 2017). Blimps may offer a non-lethal alternative with clear conservation benefits for target and bycatch species. Our findings also have relevance to the potential effectiveness of other emerging shark surveillance methods such as drones (Colefax et al., 2018; Kizka and Heithaus, 2018) despite their shorter battery life. We foresee an expanded role of this technology in general beach safety and risk reduction, including detecting and monitoring rips and other hazards that contribute to drowning deaths; occurrence of drowning deaths is an order of magnitude higher than shark incidents (Simmons and Mehmment, 2018). Our fusion of zeppelin-era technology with modern-day optics, communications and computing power provides a simple, environmentally sustainable and socially accepted method for improving beach safety, and could perhaps ultimately replace lethal approaches to managing risks associated with human-shark encounter.

6.5 References

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Chapter 7– General Discussion

A key aim of applied marine conservation research is to understand and moderate the impact of human activities on animal populations. Humans are impacting elasmobranch populations with declines being documented in fisheries around the world (Stevens, 2000). The full impacts of fishing induced abortion in sharks and rays has not been comprehensively studied and as a result, the potential for discarding to contribute to population impacts is unclear. The development and refinement of new sampling techniques as outlined in this thesis (e.g. stereo-BRUVs, acoustic telemetry, and affordable portable ultrasound devices) now provide the opportunity to explore these poorly defined impacts, with no-take marine reserves providing an ideal study system. Although, the issue of post-capture mortality is well established in elasmobranchs, little consideration has been given to the potentially pervasive effects of capture-induced parturition on discarded elasmobranchs. Bycatch and discarding is particularly high in gill-net fisheries, with the success of bycatch mitigation strategies dependant on species and location (Waugh et al., 2011). Gill-nets are deployed extensively in a number of countries as a strategy to cull sharks for the protection and safety of bathers, but have high levels of bycatch. Managers have few options to reduce this bycatch and there are few socially accepted or proven alternative methods to mitigate shark incidents. Quantifying the full impact of fishing on discarded species and providing means to reduce bycatch and discarding is likely to improve the management of elasmobranchs as a component of biodiversity.

I aimed to address a number of these knowledge gaps by using of a wide variety of methods and developing new techniques. First, using a systematic review I define, examine and present analyses on the prevalence of capture-induced parturition. I sought to determine the species and species traits that make elasmobranchs most susceptible to this phenomenon (Chapter 2). I explore the movement patterns and residency of this commonly discarded elasmobranch to better assess habitat requirements and space use in relation to marine protected areas (Chapter 3). I then expand our knowledge on the impacts of capture-induced abortion by comparing the abundance, size and pregnancy rates of a commonly discarded elasmobranch in fished and no-take areas within a marine protected area (Chapter 4). In Chapter 5 I introduce the seemingly disparate technique of aerostats as an ecological sampling technique which provide continuous monitoring of the nearshore environment with potential for offering an observational alternative to shark-

nets as a shark-mitigation technique. Chapter 6 solidifies the application of this technique for detecting sharks in proximity to ocean users and therefore the potential of an aerial platform (aerostat) replacing a high bycatch method. Through social surveys I quantified the response of beach-users to shark-nets in comparison to my new approach.

7.1 Capture-induced parturition: the value of model taxa

A key finding demonstrated in chapter 2 is the prevalence of capture-induced parturition by live-bearing elasmobranchs in response to various methods of capture. This potentially threatening process is rarely considered within population or fisheries models and threat assessments. This is especially concerning for species with high discard rates, with these species potentially losing considerable reproductive output. In Chapter 4 I then demonstrate that the condition can indeed affect recruitment in a commonly discarded species, and therefore can be inferred to potentially impact a substantial proportion of live-bearing sharks and rays. For discarded species, indirect impacts of fishing on recruitment are difficult to measure due to impacts spanning multiple generations. Despite this, recent advances have been made, with work by Guida et al. (2017) showing that exposure of pregnant female *Trygonorrhina dumerilii* to simulated trawls results in reduced size and weight of progeny, inferred to result in reduced fitness of their offspring. This reduced fitness is likely to be similar for those offspring experiencing premature birth due to capture as described in Chapter 2 and deserves further research. Wosnick et al., 2018 show that pregnant guitarfish *Zapteryx brevirostris* have high rates of abortion and pregnant individuals have significantly higher post-capture mortality. These findings, when considered together, demonstrate that pregnant females are disproportionately sensitive to the impacts of fishing and add support the strategy of locating and protecting nursery grounds put forward by Heupel et al. (2007).

Given the threatened nature of many sharks and rays the minimisation of impact and the development of non-lethal sampling techniques is an important area of research (Hammerschlag and Sulikowski, 2011). However, non-invasive techniques for sampling the impacts of fishing are difficult and sometimes impossible to develop because fishing is the treatment of interest. In Chapter 4 I pioneer a means to test for capture-induced abortion with minimal intervention of animals by using marine park zones with different levels of fishing as sampling locations. Most estimates of abortion in elasmobranchs are from studies focused on better understanding demography, such as fecundity, age and maturity data (e.g. Ebert, 1984; Struthsaker, 1969; Snelson et al., 1988). The collection of these data usually requires dissection of animals and therefore lethal sampling. New techniques have recently been developed, however, that may

allow suspected abortion events to be identified without the need to euthanize and dissect animals. Wosnick et al., (2018a) present a new method for diagnosing the occurrence and frequency of suspected abortion events via ultrasound. Even for future studies that employ this new method, the fact that abortion rates are species and fishing method specific, means it would quickly become prohibitively costly to estimate abortion rates for every discarded species. Therefore it is imperative to develop our understanding and monitor the impact of discarding using model or indicator species.

It can be a challenge collecting population data and measuring impacts on species that are rarely encountered. This is often the case with elasmobranch taxa and hence there is a need to test hypotheses on model/indicator species that are ubiquitous with findings able to be generalised to inform the management of less common species. In this context, indicator species refer to species whose abundance might provide a guide to levels of fishing disturbance and therefore the level at which communities in an area being impacted by fishing (Macdonald, 1996). Ideally indicator species, are ubiquitous but sensitive, with measurable responses to disturbance in terms of abundance (Macdonald, 1996). Indicator species are typified by being particularly sensitive to impacts due to being slow growing and having relatively low recruitment, and to be useful indicators they need to be widespread and easily recognized (Macdonald, 1996). Using a commonly discarded elasmobranch *Trygonorrhina fasciata* as a model species in chapter 4 I demonstrated that discarding has impacts beyond post-capture mortality. Given that there is limited information on the biology, abundance and populations status of many sharks and rays I provide evidence that *Trygonorrhina fasciata* can potentially be used as an indicator species to detect impacts of fishing. Sharks and rays are known to be sensitive to fishing impacts with *Trygonorrhina fasciata* provide a good model and potential indicator species as they are relatively abundant (Fetterplace, 2018) and easily identified. They are also likely to show a growth rate common to many threatened species (Dulvy et al., 2014). The maximum intrinsic rate of population increase of banjo rays (D’Alberto et al., 2019) has been found to be similar to that of sawfish species (Family Pristidae), which are the most threatened family of elasmobranch (Dulvy et al., 2014). Rhinobatidae and Trygonorrhinidae (banjo rays) are among those species known for their relatively low intrinsic rate of population increase (D’Alberto et al., 2019.). Their large body size, slow life history make their response to impacts rapid and declines readily apparent (Stevens, 2000). All of these species groups use inshore habitat in heavily fished coastal regions making them vulnerable to impacts (D’Alberto et al., 2019). Keeping this in mind, species like *Trygonorrhina fasciata* may prove a useful indicator of fishing impacts, such as capture-induced

parturition, which are likely to impact other species in a similar manner although impacts are much harder to detect in rarer species. Given their abundance within a marine protected area, *T. fasciata* offered the unique opportunity to quantify the impact of capture-induced abortion that would be difficult for less abundant elasmobranchs due to infrequent encounters.

7.2 Bycatch reduction through gear modification or elimination

In Chapters 2, 3 and 6 I present a number of reasons that bycatch of sharks and rays needs to be measured, managed and ideally reduced. Options do exist for reducing bycatch through gear modification or elimination. Many gear modification techniques utilise species traits to reduce catches of specific species. For example ‘pingers’ are often placed on nets for deterring echolocating species (Mackay and Knuckey, 2013), mesh size is chosen to exclude species or individuals based on size (Faife, 2003). While gear modification may work for one species or species group, it may be ineffective for another (Waugh et al., 2011) and has even been shown to reduce the catch of target species in some circumstances (Melvin et al., 1999). It is therefore unlikely that gear modification alone can be used to effectively mitigate the impact of fishing on bycatch species. The other option for mitigating bycatch impacts is elimination. This can be achieved by a number of means, including by establishing marine parks or removing gear from the water. Removing fishing gear is often contentious and often involves commercial fishing licence buybacks or, as is the case for shark-nets outlined in Chapter 6, by finding alternative means of achieving the same goal (human safety) without the need for nets in the water (Chapters 5 and 6). Fishing buy backs are a strategy aimed at combating overfishing of targeted stocks but generally have mixed success due to the presence of unused licences, effort creep, and negative social consequences (Clark et al., 2005; Teh et al., 2017). Elimination via my proposed use of blimps as an alternative method to provide human safety outcomes is also an isolated case and this method of bycatch reduction is only applicable to shark-nets specifically. With this in mind, a promising method for managing bycatch issues in elasmobranchs is adequately sized, placed and enforced marine protected areas as detailed in Chapter 2, 3 and 4.

7.3 Conservation value of spatial closures: no-take marine reserves

The value of marine protected areas can extend beyond pure conservation objectives, with the use of these areas for research enabling hypotheses to be tested that would otherwise go unanswered. Using a marine protected area as a model system in Chapter 4 I demonstrate a previously undescribed impact of fishing on elasmobranchs. In addition to conserving biodiversity, marine protected areas are known to provide an opportunity to gain a better understanding of the dynamics of intact marine systems (Gaines et al., 2010). No-take zones offer

an opportunity to detect effects previously not considered by providing reference areas with low anthropogenic impacts. The apparent decrease in recruitment in fished areas I presented in chapter 4 would not have been detected if reference locations where fishing does not occur were not available. Such areas provide a baseline of relatively pristine systems without which the impacts occurring to other areas may go unnoticed (Knowlton and Jackson 2008). By using MPAs as an unfished standard, Edgar et al., 2014 show considerable fishing effects on temperate and tropical reef fish communities. They show that shark numbers were greatly reduced in fished areas globally, with 93% of sharks, (and 63% of all fish biomass) removed from reefs by fishing. Notably these losses were due to targeted fishing, and not discarding, however I show in Chapter 1 and 2 the potential impacts on discarded species. I provide a clear example of the importance and value of marine protected areas, and no-take marine reserves more specifically, as research tools to identify and quantify effects of fishing on marine ecosystems.

It is established that marine protected areas need to be of adequate size, shape and position in order for species to show a response to protection (Kramer and Chapman, 1999). In particular, larger-scale MPAs may offer better potential for protecting wider-ranging species, but larger MPAs require more resources for monitoring and enforcement (MacKeracher et al., 2018). It is recognized, however, that most MPAs have been established for reasons other than solely protecting sharks and rays (MacKeracher et al., 2018). Therefore, they may not necessarily be adequately sized or placed to protect elasmobranch species. In some cases, however, the conservation of elasmobranchs can be facilitated in by the use of marine protected areas, especially species that show residency or that use specific nursery, reproduction or feeding areas (Speed et al., 2016). I provide a robust example of a resident elasmobranch species showing a response to protection by examining residency and movement patterns (Chapters 3 and 4) in conjunction with abundance. Data such as these are of value to management with Heupel and Simpfendorfer (2005) suggesting that the amount of time that species spend within protected-area boundaries is a key criteria for successful reserve design. Given that the ray species I have focussed on is predominantly found on soft-sediment habitats (Rees, 2017) and species on soft sediment have generally thought to not exhibit residency to a specific site (Caveen et al., 2012) this high level of residency could be considered surprising. My work in chapters 3 and 4 adds to emerging research, however, that shows that species on soft-sediment can indeed show site attachment and residency (Fetterplace, 2014; Adams, 2016; Fetterplace, 2018) and therefore may respond to spatial management.

Given social and economic considerations (Watson et al., 2014; MacKeracher et al., 2018), spatial closures such as no-take marine reserves are not always possible and are not necessarily the most appropriate option for conservation. In cases where permanent reserves are not applicable, seasonal closures protecting nursery areas or aggregation sites could offer temporary spatial refuge for species during critical reproductive periods (Bensley et al., 2010). Whether such strategies can be broadly applied to all elasmobranchs is still debated given that not all species have clearly defined nursery areas and many range broadly (Heupel et al., 2007). Many species show philopatry, however, with individuals returning to specific sites for reproduction or foraging demonstrated to occur in numerous elasmobranch taxa (Chapman et al., 2015). These studies offer support to the notion that such species may benefit from at least seasonal protection. It is recommended that partial-protection measures such as seasonal closures should be applied in conjunction with other management strategies which may include limiting fishing through size or bag limits to protect individuals throughout their life history (Speed et al., 2016). With the evidence presented in this thesis, placed in the context of existing literature, I suggest that no-take marine reserves can potentially help mitigate the impacts of fishing on discarded bycatch species, as long as they spend time within them during critical periods.

In addition to the size and placement of no-take marine reserves, the age of reserve can influence the response of species to protection (Edgar et al., 2014). Research suggests that older reserves appear to be more effective than younger reserves but recovery of targeted species can occur quite quickly. Abundances of directly exploited species have been shown to increase relatively rapidly, first appearing within an average of 5 years after protection (Babcock et al., 2010). Speed et al. (2016) who found evidence of population recovery in grey reef sharks *Carcharhinus amblyrhynchos* after 8 years of protection. I present data showing the recovery of the incidentally caught *T. fasciata* in response to protection, which occurred after 14 years. Although at different trophic levels, these two species have surprisingly similar reproductive parameters (Smale, 2009; Huveneers, 2015) so the rate of recovery could be expected to be similar. Barrett et al. (2007) suggest that responses to protection can be slow, complex and species-specific. Further, Babcock et al. (2010) noted that although targeted species show rapid responses, species that are indirectly impacted can take longer to show a response to protection. The fact that *T. fasciata* take longer to recover once protected might indicate that capture-induced abortion has a more subtle impact as it affects recruitment but not necessarily adult survival. It would be interesting to conduct modelling to test the estimated rate of recovery versus the rate observed in chapter 4, but that is beyond the scope of this thesis.

7.4 Future directions and concluding remarks

This thesis not only improves our understanding of the impacts of humans on the marine environment, but also explores potential solutions and I highlight several promising directions for future research. My work using an indicator species highlights the need for targeted study on the impacts of capture-induced parturition in other locations and for other species. More broadly, targeted research is needed to determine the full impact of commercial and recreational fishing on discarded elasmobranchs worldwide and to further develop mitigation strategies. Marine protected areas offer one management strategy and with the increasing technological advances in remote aerial video-surveillance information on animal abundance, movements, behaviour and ecology is becoming increasingly accessible which in turn enables better reserve design. Together, this information if incorporated into management practices will result in reduced impacts and the maintenance of ecological processes.

I demonstrate differences in recruitment for *T. fasciata* between predominantly recreationally fished areas and areas without fishing, but the impact of commercial fishing on discarded species, particularly trawling remains uncertain. It is important to recognise that fiddler rays are still reproducing in the fished areas of our study, just at a reduced rate. This indicates that although recreational fishing is having an impact, fishing effort at our study location is not at a level that is causing measurable population declines. What remains uncertain, however, is the impact that commercial and recreational fishing have in combination, particularly given that in Australia, Recreational and Commercial catches can be at similar levels (Evans et al., 2016). Trawl fishing has been shown to significantly alter the relative abundances of elasmobranchs caught as both target and bycatch species (Graham et al., 2001, Kennelly 1995). Graham et al., 2001 highlight that most trawlable ground (i.e. soft sediment) on the slope off NSW is regularly fished, which historically has resulted in fairly constant fishing pressure on the sharks and rays inhabiting these areas. This sustained trawling is the most likely cause of the 80% decrease in relative abundance of sharks and rays over 20 years on the NSW upper slope observed by (Graham et al., 2001). The potential for population impacts on *T. fasciata* and other commonly discarded species is deserving of targeted research with the small amount of evidence available suggesting that the impacts could be considerable. *T. fasciata* are regularly caught during inshore trawls with 1-5 individuals per tow (Huveneers, 2015). In comparison, trawl catch rates in unfished grounds, have been measured at 40-50 individuals per hour (Graham, pers. comm., March 2015 in Huveneers, 2015) which provides some estimate of virgin biomass. Impacts are also suggested by Fetterplace (2018) who observed an average relative abundance of 0.06 individuals per BRUV deployment

in trawled areas outside MPAs compared to 1.05 individuals within no-take areas at the same depth (60 m), and although the difference was not significant, the trend suggests that trawling may be having an impact on the abundance of this species. The impact of trawling and other commercial fishing practices known to cause stress, mortality and abortion on elasmobranchs is deserving of greater research.

Commercial and recreational fishers are not the only people to interact with elasmobranchs, and as researchers we often deliberately interact with those species that are rare and endangered. Traditional tracking techniques require invasive procedures to capture and tag the animals which we know can cause stress, mortality and abortion. In Chapter 5 and 6 I explore aerial video-surveillance as an emerging technique with great potential to provide information on animal abundance, movements, behaviour and ecology without needing to capture animals. Given the challenges associated with tracking animal movements in aquatic habitats, fine-scale movement patterns of animals in nearshore zones remain poorly researched. This lack of information is in part due to the high-energy dynamics characterising these areas, which impedes the use of acoustic or satellite tags (Raoult et al., 2018). Unlike tagging methods, aerial surveillance is a non-invasive technique and gathers continuous, direct information about animal locations and interactions (Rieucou et al., 2018). Traditional aerial surveys typically use light aircraft or helicopters, but in chapter 5 and 6 I introduce aerostats as an alternative. Using aerostats and drones as aerial platforms to conduct marine fauna surveys in place of manned aircraft is a promising area of research. Recent work highlights their use for tracking animal movements in the marine environment (Raoult et al., 2018, Rieucou et al., 2018, Ruiz-Garcia et al., in prep). Raoult et al., (2018) tracked sharks for up to 10 minutes in the shallow lagoon of Heron Island, Australia. Rieucou et al., 2018 uses drone surveys and image analysis to study shoaling behavior in reef sharks *Carcharhinus melanopterus*. Ruiz-Garcia et al. (in prep) used the aerostat technology I developed to study the fine-scale space-use of stingrays in nearshore environments. Clearly there is significant potential to improve our understanding of elasmobranch ecology and behaviour with these new technologies. In combination with my thesis, these studies highlight use of these emerging technologies and their utility for gathering information on spatial ecology in coastal zones. The use of aerial sampling is a useful tool for marine ecologists and the use of these techniques is likely to increase in the future.

I encourage research into novel techniques, including handling practices and gear modification, which may reduce the likelihood of capture-induced parturition. Better understanding of the

specific mechanisms that induce parturition on capture may also provide insight into methods to reduce its occurrence. In the meantime, general techniques to reduce stress (Cooke and Suski, 2005) should be encouraged for scientists and fishers which are covered in detail in Chapter 2. The use of more selective sampling or fishing techniques are also preferable where-by pregnant females may be avoided, for example the method of hand-capture I describe in Chapter 4. In addition, it would be useful to quantify the mortality of embryos following capture-induced parturition, to provide confirmation that capture-induced parturition contributes directly to reducing recruitment and to interpret the potential magnitude of the problem. With a better understanding of the physical and physiological mechanisms which induce parturition, techniques may be developed to reduce stress-induced parturition rates on vulnerable species, and thereby reduce the impact of catch and release angling and other fishing methods where adult elasmobranchs are discarded.

My thesis represents an important step in improving our understanding of the impacts of fishing by quantifying the prevalence and effect of capture-induced parturition on discarded elasmobranchs. I provide the first indication that capture-induced parturition can lead to population level effects with measurable impacts on recruitment. Furthermore, I present evidence that marine protected areas may offer a means to mitigate this impact. I also develop a new aerial surveillance method which proved to be an effective and socially preferred alternative to a high bycatch fishing method and shark mitigation strategy, thereby providing a way to reduce bycatch and the discarding of sharks, rays and other marine life. In a global context, this thesis, together with other work I have been involved with (Wosnick et al., 2018a, Wosnick et al., 2018b) represents an important contribution to the conservation and responsible management of elasmobranchs. It is envisaged that the findings of this work will be more broadly adopted to improve the management of sharks and rays as an important component of marine biodiversity.

7.5 References

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Appendix 1

Table A1.1 - Species and link to video or images depicting a capture-induced parturition event (and one stranding: #21). If the video link does not work, or a video has been removed, please contact the corresponding author and a viewing can be arranged if required.

#	Species	Link
1	<i>Aetobatus narinari</i>	https://www.youtube.com/watch?v=tvT1uLyIGN4
2	<i>Hemitrygon fluviorum</i>	https://www.youtube.com/watch?v=ysHCpH1kEfk
3	<i>Hexanchus griseus</i>	https://www.youtube.com/watch?v=b70GnqY2iQc
4	<i>Hypanus sabinus</i>	https://www.instagram.com/p/BIIX0JyDOAr/?taken-by=scottyjrfishing
5	<i>Hypanus sabinus</i>	https://www.youtube.com/watch?v=LzzCNfh6KoM
6	<i>Hypanus sabinus</i>	https://www.youtube.com/watch?v=4HWNpXiWEh8
7	<i>Hypanus sabinus</i>	https://www.instagram.com/p/sWNK6Bt1PG/?taken-by=daft_hound

#	Species	Link
8	<i>Hypanus sabinus</i>	https://www.youtube.com/watch?v=cvGqJ0_mtSo
9	<i>Hypanus sabinus</i>	https://www.youtube.com/watch?v=9ENscXNcxoc
10	<i>Hypanus say</i>	https://www.youtube.com/watch?v=ztximLv2PQ
11	<i>Hypanus say</i>	https://www.youtube.com/watch?v=qRYiEs4iVQ
12	<i>Hypanus say</i>	https://www.youtube.com/watch?v=WHUKwqJlmpg
13	<i>Hypanus say</i>	https://www.youtube.com/watch?v=5sv83S_wVRQ
14	<i>Isurus paucus</i>	https://www.youtube.com/watch?v=v6VbtMzMCoY
15	<i>Maculabatis gerrardi</i>	https://www.youtube.com/watch?v=qlt7Gtd3lOM
16	<i>Myliobatis californicus</i>	https://www.youtube.com/watch?v=p06TCTgSnWI
17	<i>Myliobatis californicus</i>	https://www.instagram.com/p/3FWcRvnwU0/?tagged=stingraybabies
18	<i>Negaprion brevirostris</i>	https://www.facebook.com/biminisharklab/videos/937696579676405/
19	<i>Pristis pectinata</i>	https://www.youtube.com/watch?v=gt-sE14dYXs
20	<i>Urogymnus polylepis</i>	https://www.youtube.com/watch?v=6YKVYJE1HY8

#	Species	Link
21	<i>Rhinobatos annulatus</i>	https://www.facebook.com/LuderitzMarineResearch/photos/a.970032346378288.1073741862.696370223744503/970033839711472/?type=3andtheater
22	Unknown	https://www.youtube.com/watch?v=7ZTrHPnCjvs
23	Unknown	https://www.youtube.com/watch?v=ZDjXS0vnr0g
24	Unknown	https://www.youtube.com/watch?v=nvF7wjzdiY
25	Unknown	https://www.youtube.com/watch?v=BjezRCPVvY4
26	Unknown	https://www.youtube.com/watch?v=e1qePnYaFK8
27	Unknown	https://www.youtube.com/watch?v=7sCV7DkEtdE
28	Unknown	https://www.youtube.com/watch?v=NJpeYTf2-O8
29	Unknown	https://www.youtube.com/watch?v=MSJzNAz1hbk
30	Unknown	https://www.youtube.com/watch?v=D1vUNOazblQ
31	Unknown	https://www.youtube.com/watch?v=PyI5JTCXoVI
32	Unknown	https://www.youtube.com/watch?v=yVETK5bhpN0
33	Unknown	https://www.youtube.com/watch?v=a0bTeKxMBM0
34	Unknown	https://www.youtube.com/watch?v=xJVHsn2CC6c

#	Species	Link
35	Unknown	https://www.youtube.com/watch?v=8Ong2v7nhBg
36	Unknown	https://www.youtube.com/watch?v=8qYMtqnQrGU
37	Unknown	https://www.youtube.com/watch?v=JntF6juiaW4
38	Unknown	https://www.youtube.com/watch?v=s7tpluNwUdA
39	Unknown	https://www.youtube.com/watch?v=0cBfTbvYxk
40	Unknown	https://www.youtube.com/watch?v=SXkzlyo77Ho
41	Unknown	https://www.youtube.com/watch?v=r5Z1GD3FSNU
	Re-upload of previous video	Re-upload of previous video
42	<i>Hypanus sabinus</i>	http://imgur.com/gallery/M9EMyUe
43	Unknown	https://www.youtube.com/watch?v=X5Ke352_F_8
44	Unknown	https://www.youtube.com/watch?v=l7M5w_gkC1E

Table A1.2 Species classification, fishing method used and number of embryos resulting from capture induced parturition (and one stranding: #21). The title of the video/images is also given.

#	Order	Family	Method	Number of embryos seen in video (or reported by uploader)	Title
1	Myliobatiformes	Aetobatidae	Unknown	4	Animals 2016 - Fisherman Helps Stingray Give Birth - Stingray Giving Birth #2
2	Myliobatiformes	Dasyatidae	Unknown	10 (12)	RARE FOOTAGE: Stingray Giving Birth Boy Delivers 12 Stingrays! HD
3	Hexanchiformes	Hexanchidae	Hook and line	15	Shark give birth in Florida
4	Myliobatiformes	Dasyatidae	Unknown	3	Well.. @huntfishwrestle@kole_reeves7 and I birthed some baby rays today (the mom and babies were released safely)
5	Myliobatiformes	Dasyatidae	Hook and line	3	Stingray gives birth to live young - Florida Fishing
6	Myliobatiformes	Dasyatidae	Hook and line	2	Caught On Camera: Fisherman Helps Stingray Give Birth
7	Myliobatiformes	Dasyatidae	Unknown	3	#stingray #stingraybirth#yup
8	Myliobatiformes	Dasyatidae	Hook and line	4	Stingray gives live birth !!

#	Order	Family	Method	Number of embryos seen in video (or reported by uploader)	Title
9	Myliobatiformes	Dasyatidae	Hook and line	3	Stingray Gives Birth On Land After Being Caught By Fishman
10	Myliobatiformes	Dasyatidae	Hook and line	2	Sting Ray Birth on Fishing Line
11	Myliobatiformes	Dasyatidae	Hook and line	2	Man catches stingray while it's giving birth..unhooked and released...!
12	Myliobatiformes	Dasyatidae	Hook and line	3	Stingray Gives Birth On Beach Emerald Isle North Carolina July 2014
13	Myliobatiformes	Dasyatidae	Hook and line	3	Fishermen Caught Stingray and it Gives Birth
14	Lamniformes	Lamnidae	Unknown	10	DEAD shark giving birth
15	Myliobatiformes	Dasyatidae	Hook and line	2	Sting ray giving birth..motherhood so special.
16	Myliobatiformes	Myliobatidae	Hook and line	1	Manta Ray Birth
17	Myliobatiformes	Myliobatidae	Hook and line	2 (10)	Helped stingray give live birth to 10 stingray babies. Yes

#	Order	Family	Method	Number of embryos seen in video (or reported by uploader)	Title
					we rescued them and released them back into the ocean. @rit_chac
18	Carcharhiniform es	Carcharhinid ae	Strike net	1	none
19	Rhinopristiform es	Pristidae	Longlin e	5	Andros Expedition Report: Sawfish Birth
20	Myliobatiforme s	Dasyatidae	Hook and line	2 (3)	Jeremy Wade Catches A Giant Pregnant Stingray - River Monsters
21	Rhinopristiform es	Rhinobatidae	Strandi ng	6 (8)	Birth of baby sandsharks (lesser guitarfish <i>Rhinobatos annulatus</i>).
22			Unkno wn	16 and two eggs	Shark gives birth on beach
23			Unkno wn	5	Sting Ray giving birth full footage
24			Unkno wn	3	Stingray gives birth on boat (warning: lots of foul language in the commentary)

#	Order	Family	Method	Number of embryos seen in video (or reported by uploader)	Title
25			Hook and line	1	Stingray giving birth to baby
26			Hook and line	3	Stingray giving birth1
27			Hook and line	3	The fisherman caught a giant stingray fish give birth
28			Hook and line	1 (2)	Sting Ray giving birth in Hilton Head SC
29			Hook and line	5	Sting Ray gives birth
30			Unkno wn	3	Incredible Stingray giving birth
31			Unkno wn	3	Stingray giving birth.
32			Hook and line	3	Stingrays having babies
33			Unkno wn	4	Baby stingrays born on the beach shore
34			Hook and line	3	Caught On Camera Fisherman Helps Stingray Give Birth

#	Order	Family	Method	Number of embryos seen in video (or reported by uploader)	Title
35			Hook and line	2	Pregnant stingray gives birth in front of me!
36			Hook and line	4	Team George - Stingray gives birth to 4 babies
37			Hook and line	2	Stingray gives birth on fishing boat
38			Hook and line	4	Animals 2016 - Stingray Giving Birth
39			Unknown	3	Stingray giving birth
40			Unknown	14	Stingray giving birth inside the boat
41			Hook and line	1	Shark Gives Birth After Being Caught
	Re-upload of previous video				
42	Myliobatiformes	Dasyatidae	Unknown	3	Stingray giving birth to triplets
43			Hook and line	2	Look at this stingray having babies

#	Order	Family	Method	Number of embryos seen in video (or reported by uploader)	Title
44			Hook and line	4	Teen boys deliver stingray mama's 4 babies

Table A1.3 The upload date and location of videos/images depicting capture-induced parturition (and one stranding: #21). Also given are the social media metrics for each video.

#	Date uploaded	Location	Views	Shares	Likes	Dislikes	Comments
1	9/08/2016	Unknown	58,615		75	23	8
2	1/12/2015	Australia	10,947,946		96,404	6057	14,531
3	22/01/2016	Florida	4906		12	6	5
4	1/08/2016	North Carolina	372,906		5381		1376
5	6/07/2015	Florida	2894		8	0	2
6	22/07/2014	America	21,152,763		56,194	2708	4443
7	31/08/2014	America	unknown		23		29
8	16/08/2014	Gulf coast in Biloxi Mississippi	624,673		1256	279	209
9	23/08/2011	Unknown	2,083,896		3166	11,664	6134
10	3/08/2014	Hilton Head Island in	331		3	0	0

#	Date uploaded	Location	Views	Shares	Likes	Dislikes	Comments
		South Carolina					
11	25/06/2014	St. Augustine Florida	1,850,704		4321	654	815
12	29/10/2014	Emerald Isle, North Carolina	3434		14	3	5
13	9/08/2015	Palms Beach in Charleston South Carolina	13,378		23	11	3
14	29/11/2015	Unknown	29,988		38	32	24
15	7/06/2014	Unknown	62		0	0	0
16	21/07/2010	California	79,603		52	440	158
17	24/05/2015	America	Unknown		43		4
18	18/04/2016	Bimini, Bahamas	4471	29	125		0
19	29/12/2016	Andros, Bahamas	6411		17	0	5
20	18/08/2015	Thailand	216,877		739	12	59
21	4/04/2016	First Lagoon, Luderitz, Namibia	Unknown	8192	5900		364
22	24/01/2016	unknown	639,762		1984	822	1088

#	Date uploaded	Location	Views	Shares	Likes	Dislikes	Comments
23	9/09/2009	Nansemond River, Suffolk, VA	15,510		6	36	28
24	10/10/2014	Unknown	148,450		321	154	76
25	26/05/2009	Unknown	10,672		6	12	12
26	23/07/2009	Unknown	2508		3	1	2
27	30/12/2013	Unknown	11,647		19	15	8
28	13/08/2009	Hilton Head Island in South Carolina	41,662		23	6	9
29	7/07/2014	Unknown	247		2	0	1
30	28/07/2014	Unknown	1491		10	4	1
31	9/07/2015	Unknown	223		1	0	0
32	20/07/2013	Unknown	8372		13	5	4
33	10/07/2016	Unknown	671		5	0	1
34	13/10/2016	Unknown	14,844		31	2	2
35	23/07/2012	Unknown	3222		7	10	4
36	12/07/2013	UAE	2898		8	1	2
37	29/07/2011	Unknown	34,782		53	52	19
38	12/07/2016	Unknown	153,088		282	52	85

#	Date uploaded	Location	Views	Shares	Likes	Dislikes	Comments
39	16/06/2008	Unknown	904,741		931	159	140
40	26/12/2015	Unknown	2,433,953		8005	1668	2684
41	15/09/2013	Unknown	4471		5	17	11
	Re-upload of previous video						
42	16/08/2016	North Carolina	1491 490		4349	91	346
43	23/08/2013	Unknown	4426		13	4	2
44	25/06/2013	Unknown	8467			38	4

Table A1.4 The uploader, website and taxonomy (i.e. shark or ray) of the animals depicted in videos/image series showing capture-induced parturition (and one stranding: #21).

Number	Uploader	Website	Shark or ray
1	FUN	Facebook	Ray
2	Miller Wilson	Facebook	Ray
3	Manny A	Youtube	Shark
4	scottyjrfishing	Youtube	Ray
5	JOSE LIKES FISHING	Youtube	Ray
6	Barcroft TV	Youtube	Ray
7	daft_hound	Youtube	Ray
8	Rhonda Robbins	Youtube	Ray
9	LetyouTellit	Youtube	Ray

Number	Uploader	Website	Shark or ray
10	JWBrandon1	Youtube	Ray
11	rai martinez	Instagram	Ray
12	Wyatt Marks	Instagram	Ray
13	Ryan Copeland	Instagram	Ray
14	funny videos	Youtube	Shark
15	jamal koly	Youtube	Ray
16	ummidontnoe	Youtube	Ray
17	kahchao	Youtube	Ray
18	Bimini Biological Field Station - Sharklab	Youtube	Shark
19	Field School	Youtube	Ray
20	River Monsters	Youtube	Ray
21	Lüderitz Marine Research	Youtube	Ray
22	Bruce Leeroy Maurice	Youtube	Shark
23	M1keyDank	Youtube	Ray
24	TheBillSwerski	Youtube	Ray
25	hrmcdowell	Youtube	Ray
26	stingray129	Youtube	Ray
27	bestvines2014	Youtube	Ray
28	Bob Schatz	Youtube	Ray
29	Brooke Barraclough	Youtube	Ray
30	Rebeca Garcia	Youtube	Ray
31	James Anderson	Youtube	Ray


Number	Uploader	Website	Shark or ray
32	Rene Herrera	Youtube	Ray
33	Roberton04	Youtube	Ray
34	Hafidz Nugroho	Youtube	Ray
35	HD Gaming	Youtube	Ray
36	Vinod George Rebeiro	Youtube	Ray
37	John Moriarty	Youtube	Ray
38	FUN	Youtube	Ray
39	pengfli2008	Youtube	Ray
40	Javier Capello	Youtube	Ray
41	fireman7753	Youtube	Shark
	Re-upload of previous video	Re-upload of previous video	Re-upload of previous video
42	itsfoine 	Imgur	Ray
43	funny2me	Youtube	Ray
44	Donna Lucarelli	Youtube	Ray

Table A1.5 The total number of species and number observed to exhibit capture-induced parturition in each IUCN category from the following 12 orders of elasmobranch; Hexanchiformes, Echinorhiniformes, Pristiophoriformes, Squatiniformes, Squaliformes, Heterodontiformes, Orectolobiformes, Lamniformes, Carcharhiniformes, Torpediniformes, Rhinopristiformes and Myliobatiformes. The 343 species in the Chimaeriformes and Rajiformes orders were not included due to the results of the order level analysis (Fig. 2.4a) and to provide a more accurate estimate of expected IUCN frequencies. Seventy-eight species are Not Evaluated and the remaining 773 species from the IUCN red list can be found at the following

link:<http://www.iucnredlist.org/search/link/593e31a2-4b3dc58d>.

IUCN category	Number of species (% of total)	Number of species observed to exhibit capture-induced parturition
Least Concern	201 (23.6)	24
Near Threatened	93 (10.9)	20
Vulnerable	103 (12.1)	15
Endangered	38 (4.5)	3
Critically Endangered	17 (2.0)	4
Data Deficient	321 (37.7)	17
Not Evaluated	78 (9.2)	5
Total	851	88

Appendix 2

Table A2.1 – Tag details for the 35 rays *Trygonorrhina fasciata* with surgically implanted acoustic transmitters

Individual	VUE Tag ID	Sex	Total Length (mm)	Tagging site	Tag Family	Deployment date	Estimated tag life (days)
<i>T. fasciata</i> fish 1	A69-1601-10871	F	760	Hare Bay	V9-2x	3/03/2015	665
<i>T. fasciata</i> fish 2	A69-1601-10873	M	660	Hare Bay	V9-2x	4/03/2015	665
<i>T. fasciata</i> fish 3	A69-1601-10872	M	700	Hare Bay	V9-2x	6/03/2015	665
<i>T. fasciata</i> fish 4	A69-1601-10870	F	920	Hare Bay	V9-2x	11/03/2015	665
<i>T. fasciata</i> fish 5	A69-1601-10875	F	890	Hare Bay	V9-2x	11/03/2015	665
<i>T. fasciata</i> fish 6	A69-9002-8780	M	770	Hare Bay	V13AP	24/04/2015	155
^ Pressure sensor	A69-9002-8781						
<i>T. fasciata</i> fish 7	A69-1601-57300	M	770	Hare Bay	V13-1x	7/12/2015	904
<i>T. fasciata</i> fish 8	A69-1601-57301	F	910	Hare Bay	V13-1x	7/12/2015	904
<i>T. fasciata</i> fish 9	A69-1601-57302	M	725	Hare Bay	V13-1x	7/12/2015	904
<i>T. fasciata</i> fish 10	A69-1601-57303	M	580	Hare Bay	V13-1x	7/12/2015	904
<i>T. fasciata</i> fish 11	A69-1601-57304	F	685	Hare Bay	V13-1x	9/12/2015	904
<i>T. fasciata</i> fish 12	A69-9002-8786	F	765	Hare Bay	V13AP	9/12/2015	155
^ Pressure sensor	A69-9002-8787						
<i>T. fasciata</i> fish 13	A69-9002-8772	M	745	Hyams	V13AP	11/12/2015	155
^ Pressure sensor	A69-9002-8773						
<i>T. fasciata</i> fish 14	A69-9002-8770	F	705	Hyams	V13AP	11/12/2015	155
^ Pressure sensor	A69-9002-8771						
<i>T. fasciata</i> fish 15	A69-9002-8768	F	720	Hyams	V13AP	11/12/2015	155
^ Pressure sensor	A69-9002-8769						
<i>T. fasciata</i> fish 16	A69-9002-8766	F	755	Hyams	V13AP	11/12/2015	155
^ Pressure sensor	A69-9002-8767						

T. fasciata fish	A69-9002-8764	M	765	Hyams	V13AP	11/12/2015	155
^ Pressure sensor	A69-9002-8765						
T. fasciata fish	A69-1601-51359	M	640	Hare Bay	V13-1x	6/04/2017	904
T. fasciata fish	A69-1601-51358	F	790	Hare Bay	V13-1x	6/04/2017	904
T. fasciata fish	A69-1601-51357	F	630	Hare Bay	V13-1x	6/04/2017	904
T. fasciata fish	A69-1601-51356	M	590	Hare Bay	V13-1x	6/04/2017	904
T. fasciata fish	A69-1601-51355	F	835	Hare Bay	V13-1x	6/04/2017	904
T. fasciata fish	A69-1601-51360	F	775	Hare Bay	V13-1x	6/04/2017	904
T. fasciata fish	A69-1601-51361	F	655	Hare Bay	V13-1x	6/04/2017	904
T. fasciata fish	A69-1601-51362	F	600	Hare Bay	V13-1x	6/04/2017	904
T. fasciata fish	A69-1601-51363	F	760	Hare Bay	V13-1x	6/04/2017	904
T. fasciata fish	A69-1601-57509	M	745	Hare Bay	V13-1x	7/04/2017	1825
T. fasciata fish	A69-1601-57510	M	675	Hare Bay	V13-1x	7/04/2017	1825
T. fasciata fish	A69-1601-57511	M	675	Hare Bay	V13-1x	7/04/2017	1825
T. fasciata fish	A69-1601-51364	F	815	Hare Bay	V13-1x	7/04/2017	904
T. fasciata fish	A69-1601-57512	F	715	Bendalong	V13-1x	6/08/2018	1825
T. fasciata fish	A69-1601-57521	F	800	Bendalong	V13-1x	6/08/2018	1825
T. fasciata fish	A69-1601-57513	M	740	Bendalong	V13-1x	7/08/2018	1825
T. fasciata fish	A69-1601-57515	F	955	Bendalong	V13-1x	8/08/2018	1825
T. fasciata fish	A69-1601-57518	M	825	Bendalong	V13-1x	10/08/2018	1825

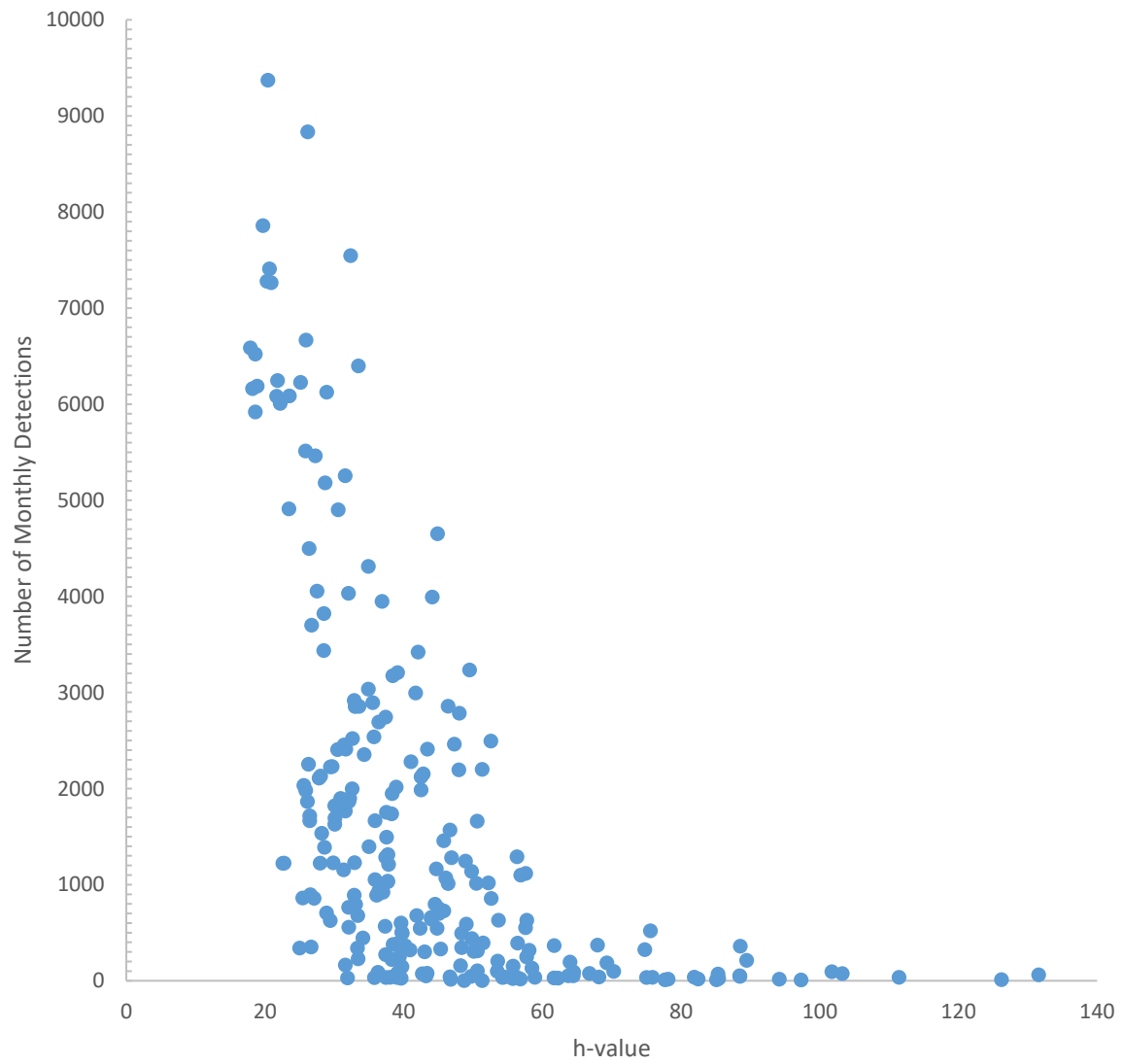


Figure A2.1 - the effect of number of monthly detections on the ad-hoc smoothing parameter (h) in the estimates of monthly KUD size in *Trygonorrhina fasciata*

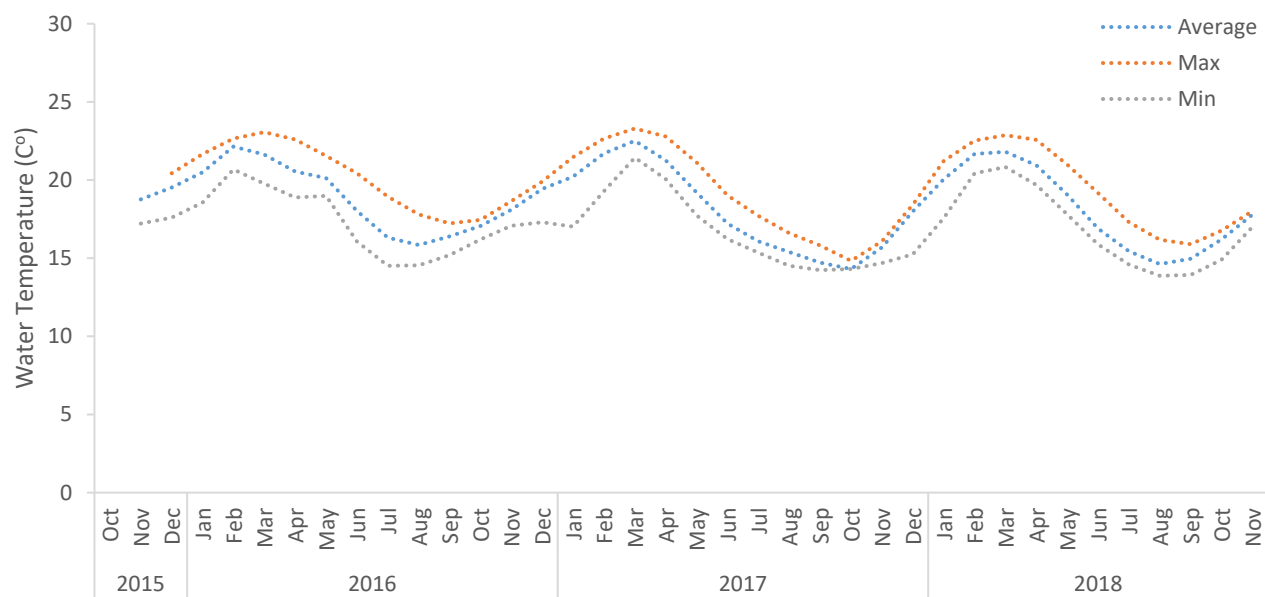


Figure A2.2 - the moving average of monthly water temperatures in Hare Bay, NSW, Australia from October 2015 to November 2018.

Appendix 3

Table A3.1 - relative abundance of fiddler rays *Trygonorrhina fasciata* between fished (HPZ) and no-take (SZ) areas within Jervis Bay Marine Park. Data were collected via stereo BRUVs across three time periods (2011 n=22 deployments, 2013 n=37 deployments and 2015 n = 43 deployments) and abundance categories consist of Juveniles (< 620 mm), Adults (> 620mm) and the sum of these two categories.

Source	df	SS	MS	Pseudo-F	P(perm)
Length Max N					
Year	2	2.2541	1.1271	3.4692	0.0368
Zone	1	2.1653	2.1653	7.7161	0.009
Location(Zone)	4	0.29999	0.92341	0.4627	0.29999
Year x Zone	2	1.7726E-2	8.8632E-3	2.7282E-2	0.9739
Pooled	91	29.564	0.32488		
Total	100	36.226			
Source	df	SS	MS	Pseudo-F	P(perm)
Juveniles					
Year	2	2.025	1.0125	3.2492	0.0441
Zone	1	5.6025	5.6025	13.964	0.0016
Location(Zone)	4	1.001	0.25024	0.37622	0.7908
Year x Zone	2	0.3722	0.1861	0.59722	0.5523
Site(Location(Zone))	6	4.1671	0.69452	2.2288	0.0485
Pooled	83	25.864	0.31161		
Total	98	40.954			
Source	df	SS	MS	Pseudo-F	P(perm)
Adults and Sub-adults					
Year	2	8.616E-2	4.308E-2	0.10979	0.895
Zone	1	0.17837	0.17837	0.59246	0.3398
Location(Zone)	4	2.8935	0.72338	1.3377	0.3589
Year x Zone	2	0.41168	0.20584	0.52458	0.5979
Site(Location(Zone))	6	3.321	0.5535	1.4106	0.2211
Pooled	86	33.745	0.39239		
Total	101	41.282			
Pairwise tests					
Length Max N					
Groups	t	P(perm)			
2011, 2013	1.2012	0.2304			
2011, 2015	2.6321	0.0089			
2013, 2015	1.8963	0.0606			
HPZ, SZ	2.7778	0.0098			
Juvenile					
Groups	t	P(perm)			
2011, 2013	0.96181	0.3417			
2011, 2015	2.1269	0.0387			
2013, 2015	2.2085	0.0329			
HPZ, SZ	3.7368	0.0024			

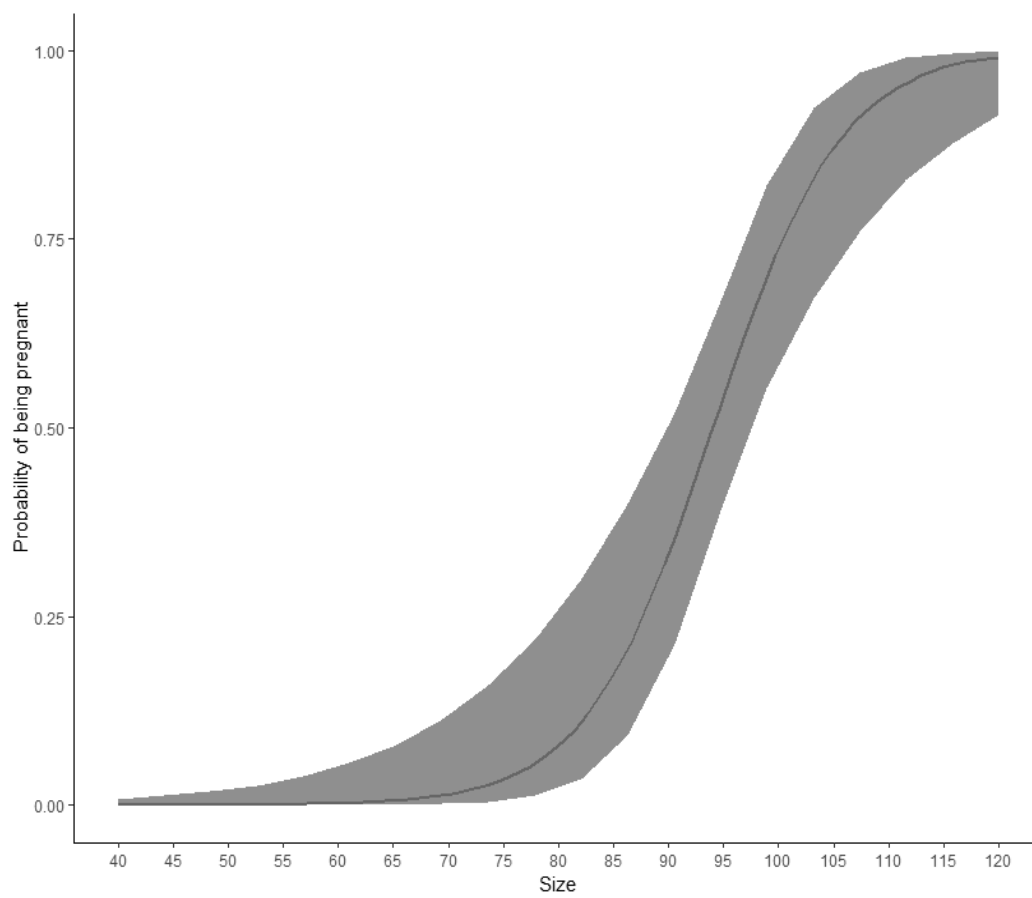


Figure A3.1 - Logistic curve \pm 95% confidence intervals describing the probability of a female fiddler ray *Trygonorhinna fasciata* being pregnant at a given length (n= 89 females).

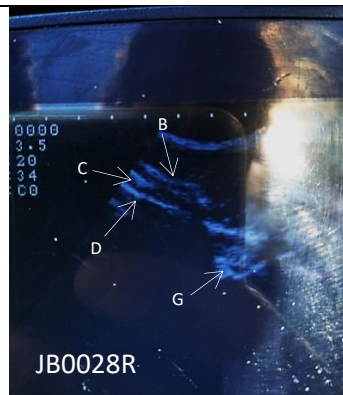
Table A4.1- Annotated ultrasound images for 103 female fiddler rays for left and right uteri. Different features are identified by:

A-Upper uterine wall,
 B-Lower uterine wall,
 C-Ventral body wall,
 D-Holding tank,
 E-Pup,
 F-Uterine fluid,
 G-Artefact,
 H-Empty uterus

M = Missing image
 0 = Not pregnant
 1 = Pregnant

External tag ID	Left Image	Right Image	Pregnant?
JB0017	M	M	M
JB0018	M	M	M
JB0019	M	M	M
JB0020	M	M	M
JB0021	M	M	M

M



JB0028

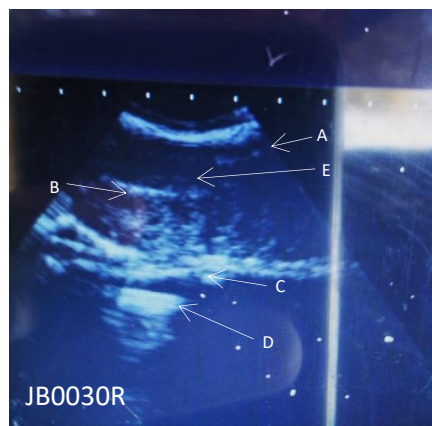
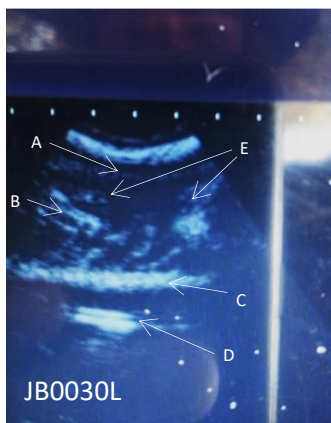
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JB0029

M

M

M



JB0030

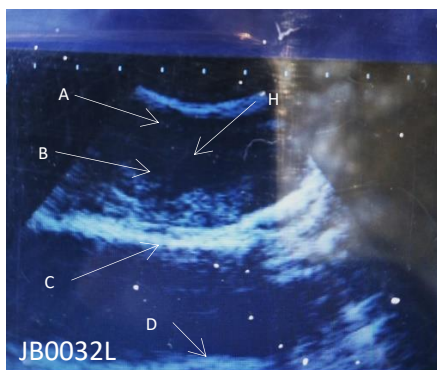
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JB0031

M

M

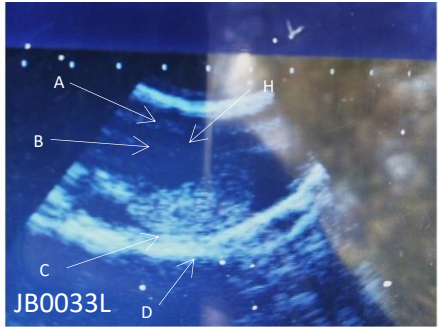
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JB0032

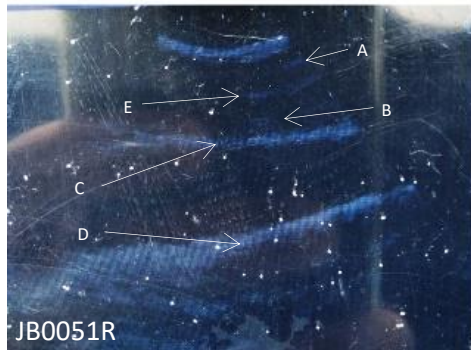
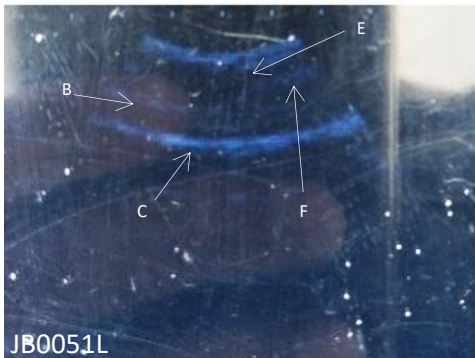


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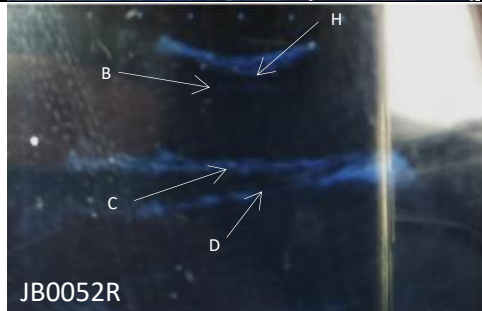
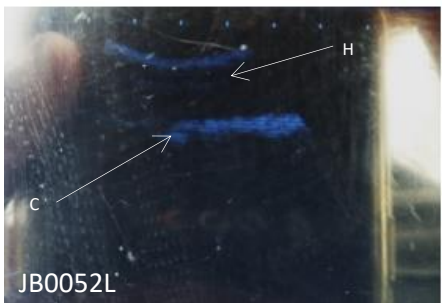


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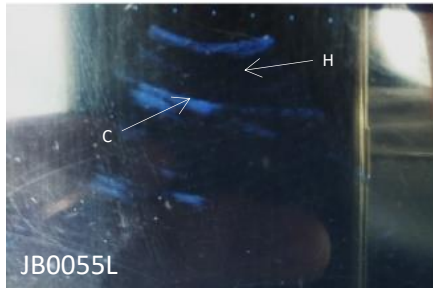
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JB0037	M	M	M
JB0038	M	M	M
JB0039	M	M	M



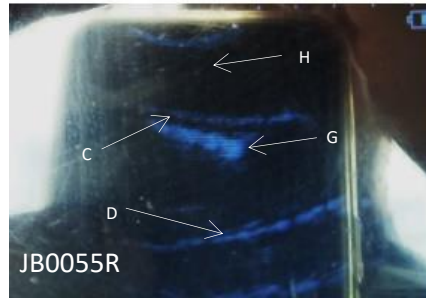
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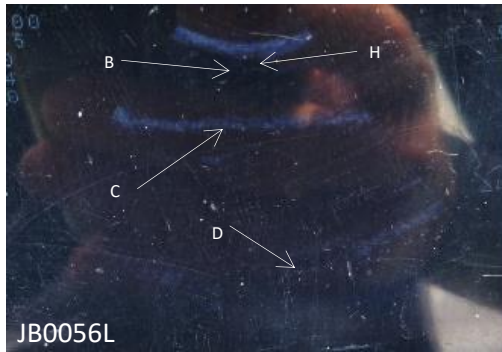
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JB0055



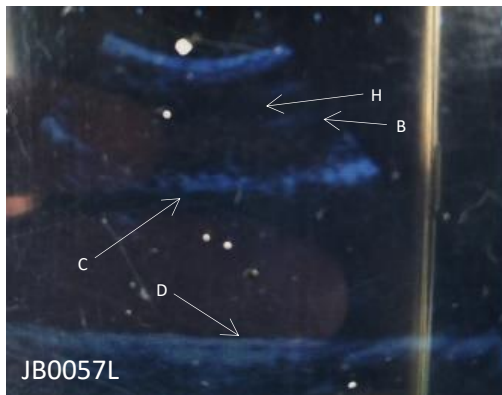
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JB0056



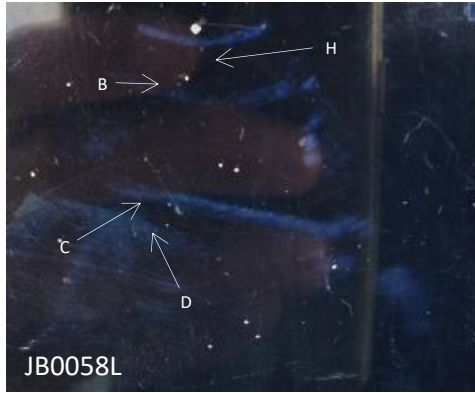
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JB0057



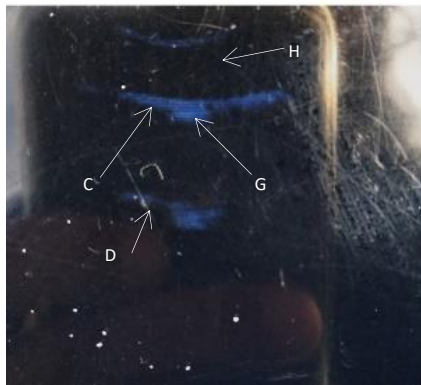
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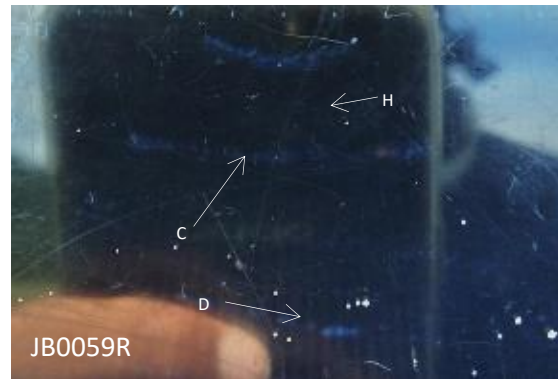
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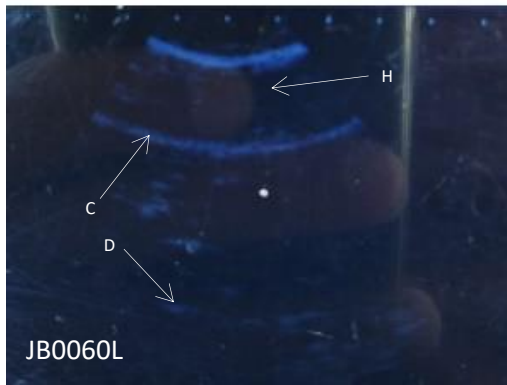
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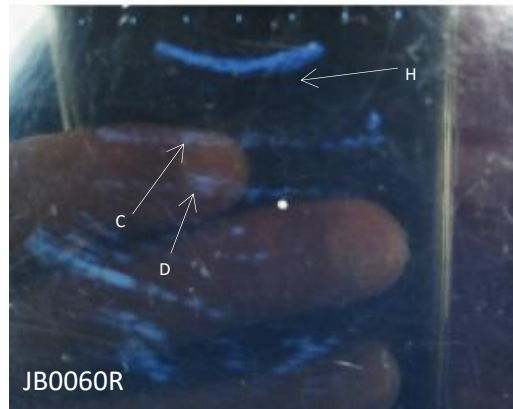
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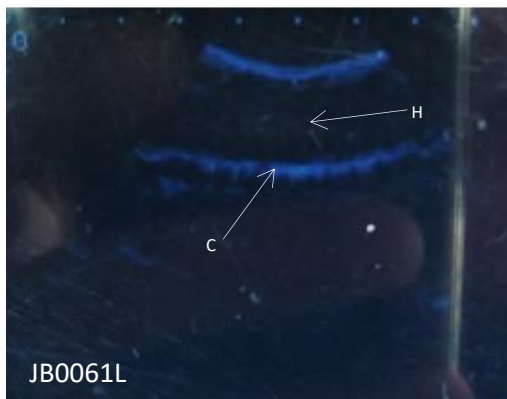


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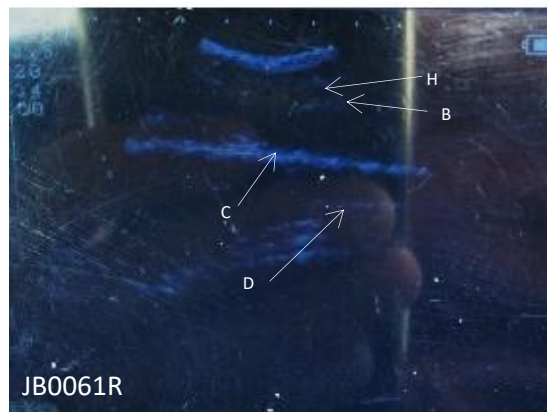


JB0060R

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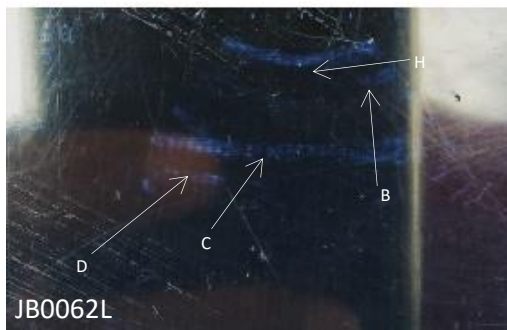


JB0061

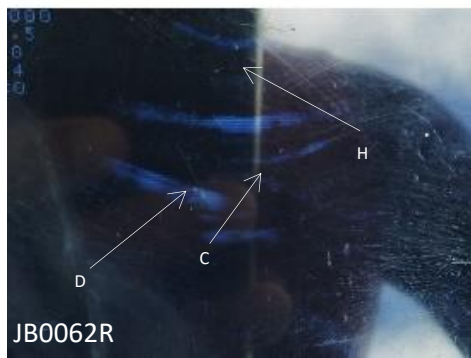


JB0061R

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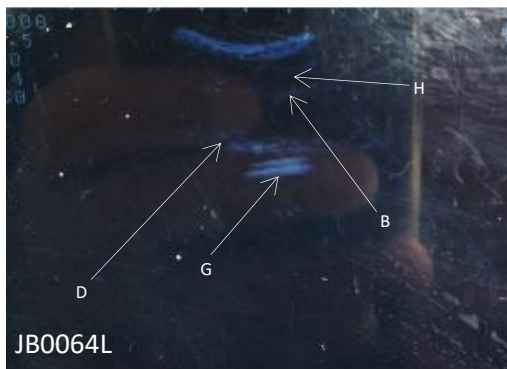


JB0062



JB0062R

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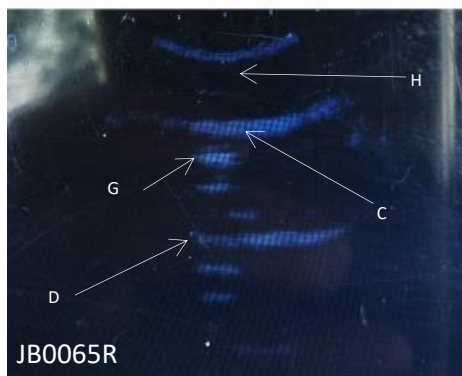
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JB0064L



JB0064R

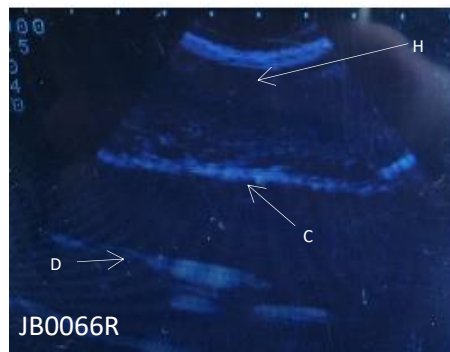
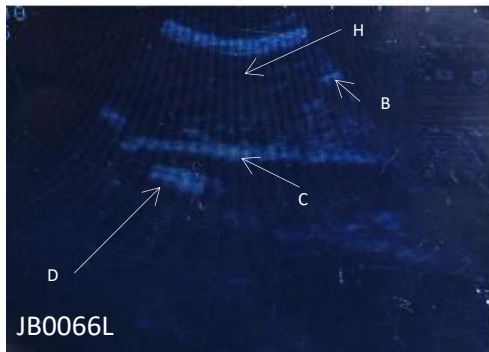
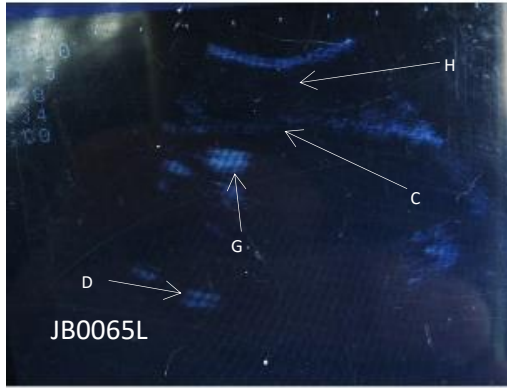
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JB0065

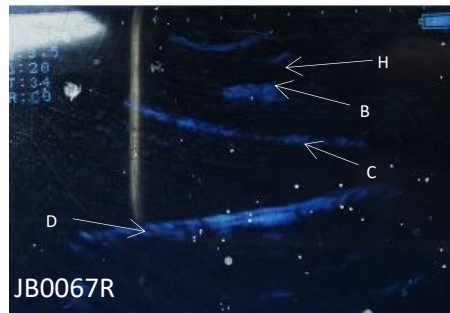
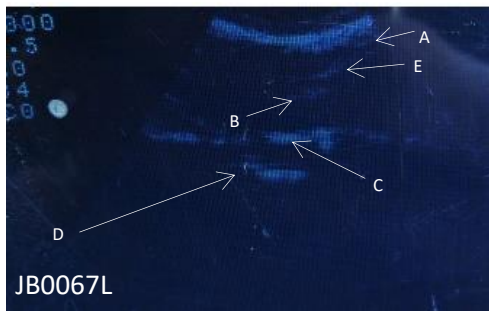
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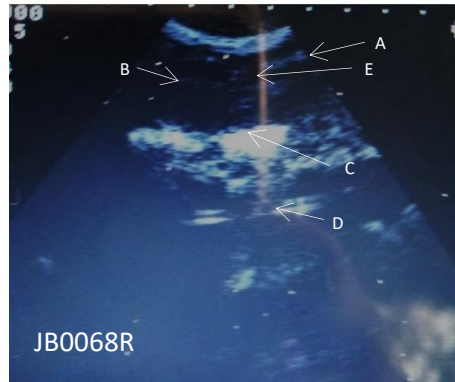
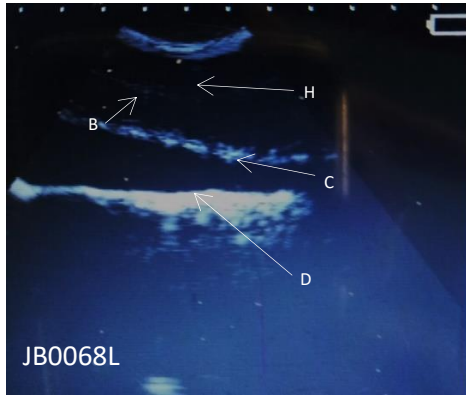
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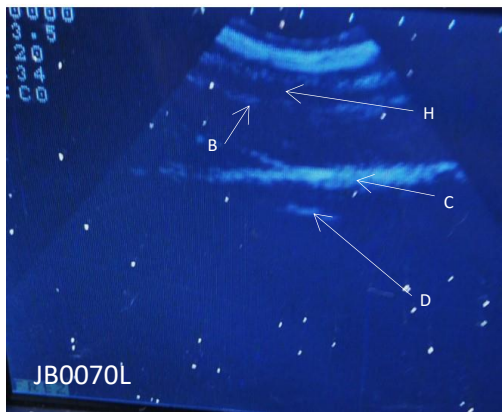
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JB0068

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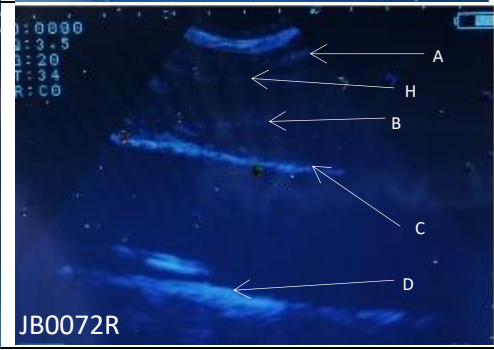
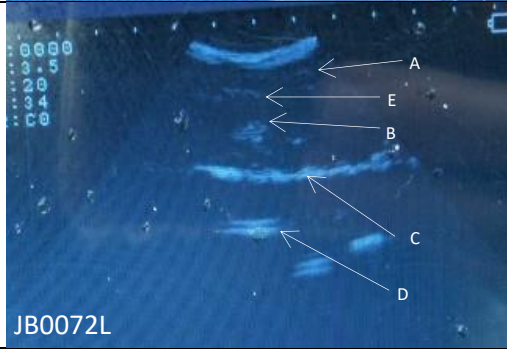
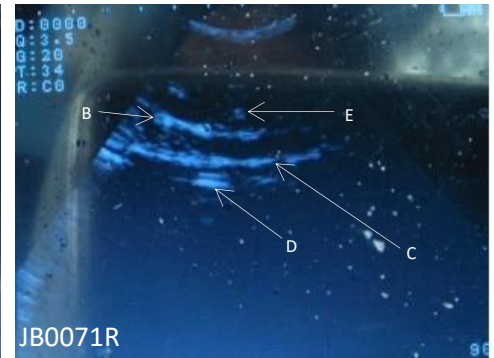
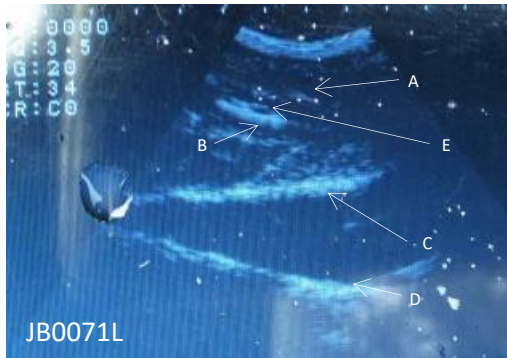
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JB0070

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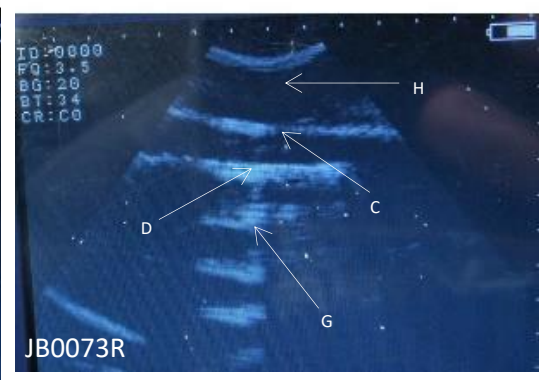
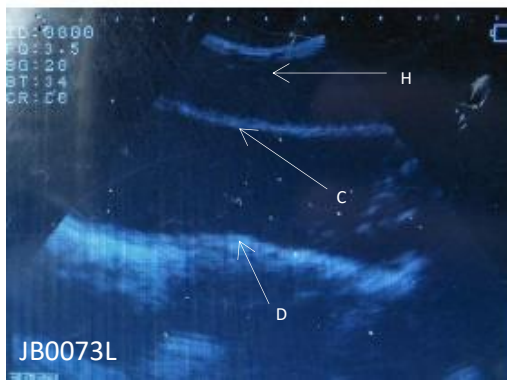
JB0071

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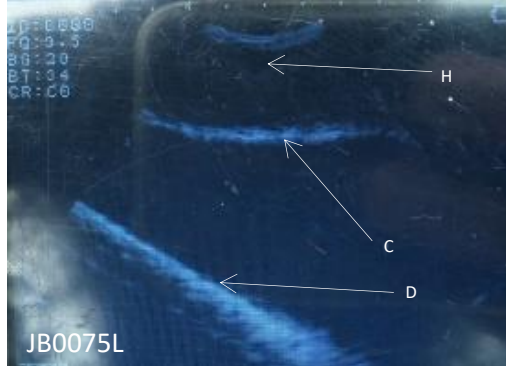
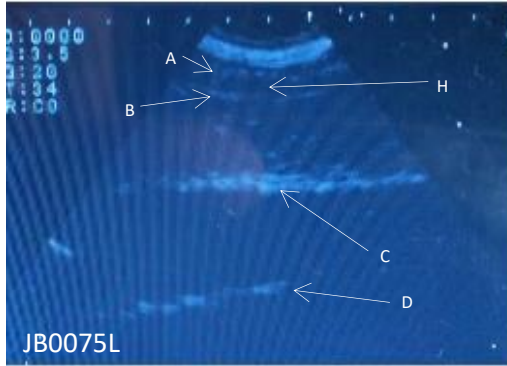
JB0072

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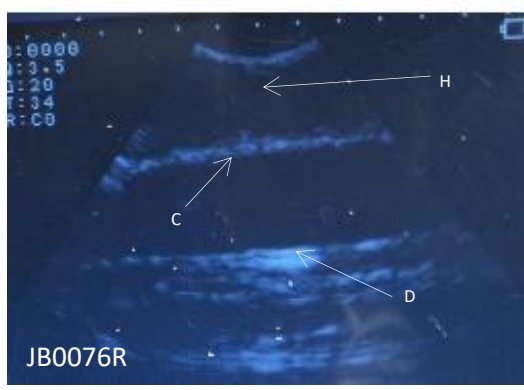
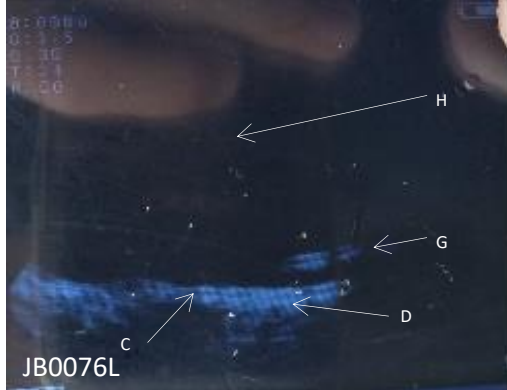
JB0073

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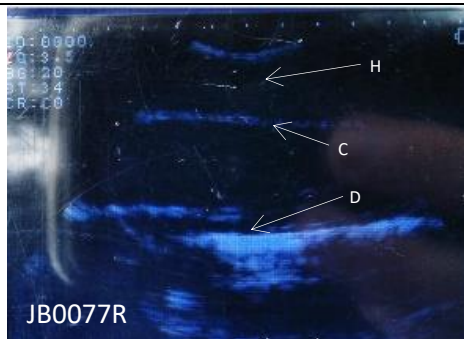
JB0075

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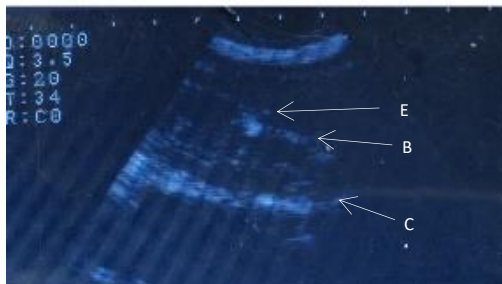

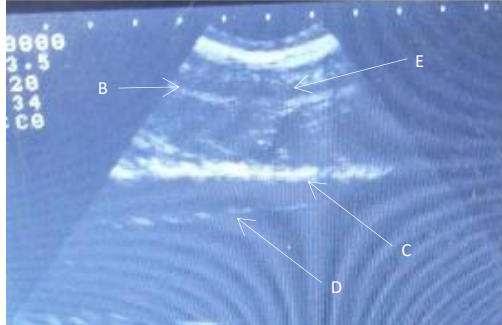
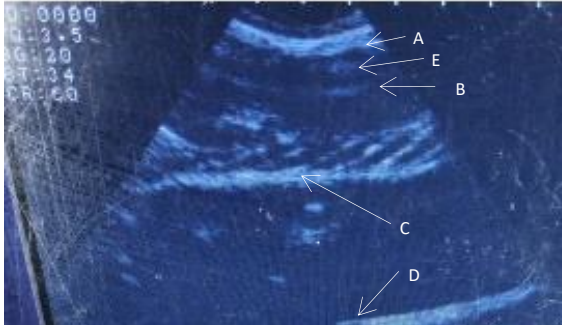
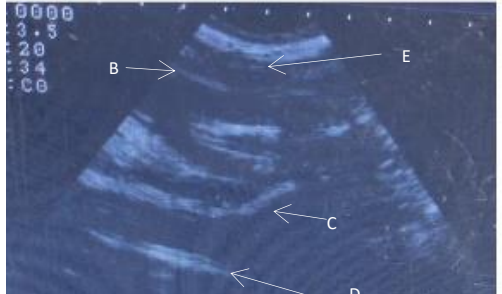
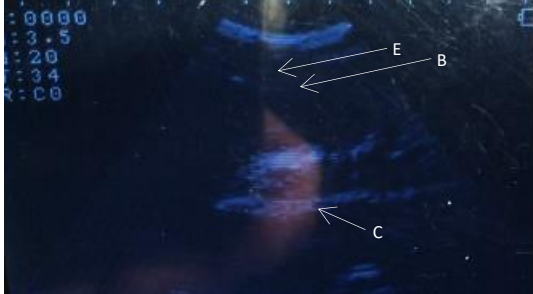
JB0076

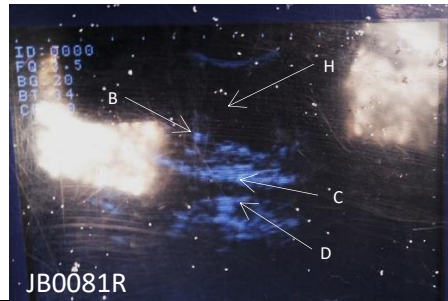
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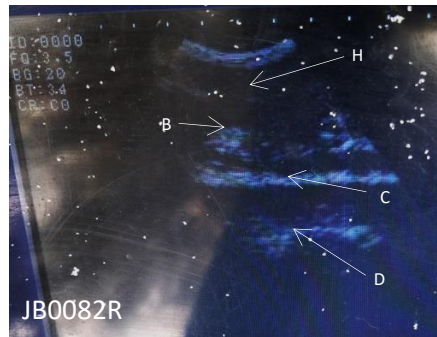
JB0077

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JB0078	 <p>JB0078L</p>	 <p>JB0078R</p>	1
JB0079	 <p>JB0079L</p>	 <p>JB0079R</p>	1
JB0080	 <p>JB0080L</p>	 <p>JB0080R</p>	1
JB0081	M		0

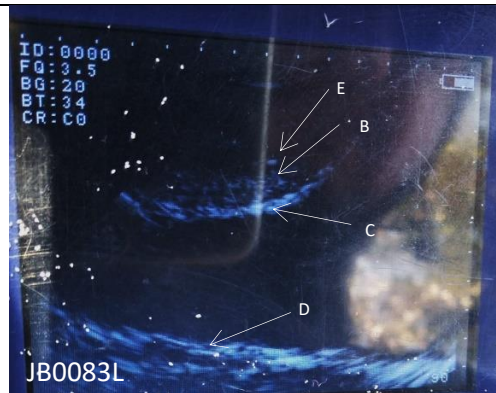


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JB0082

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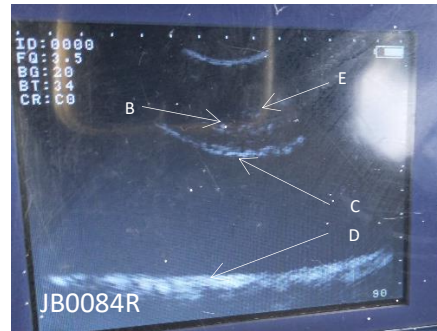


JB0083



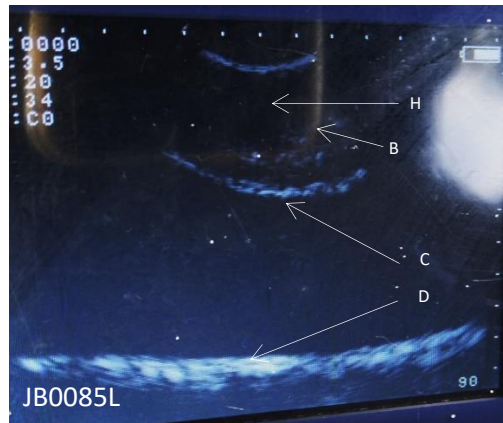
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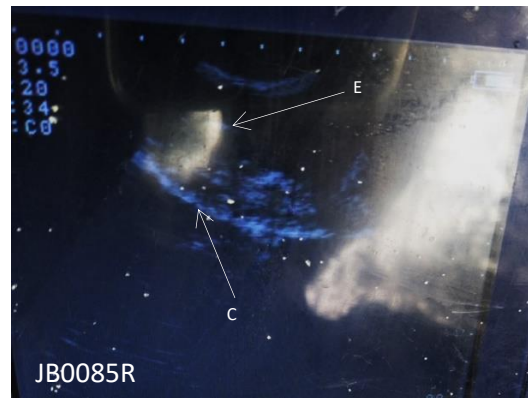


JB0084

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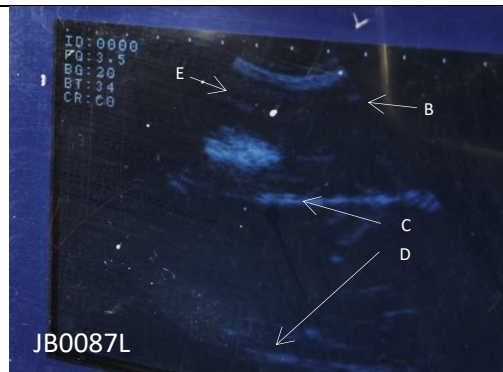


JB0085



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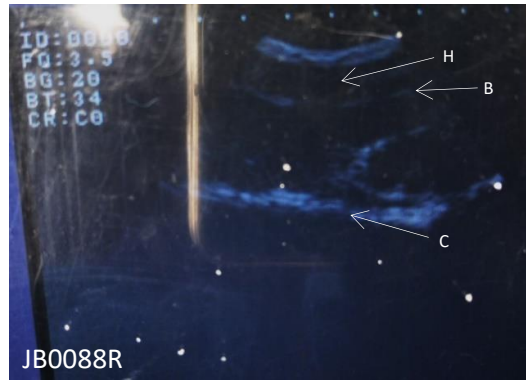
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JB0087

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M



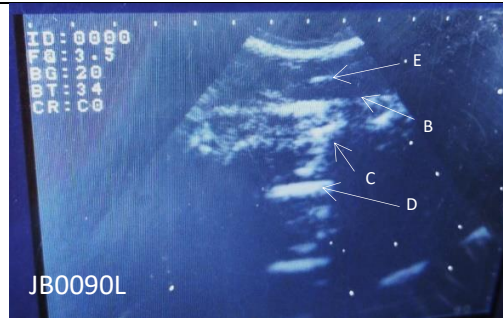
JB0088

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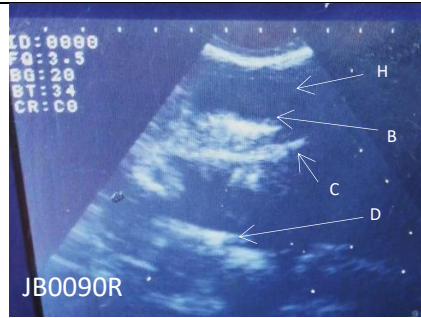
JB0089 M

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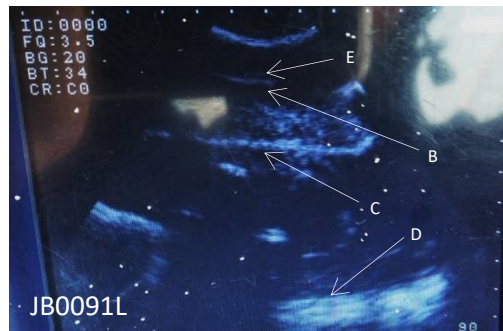
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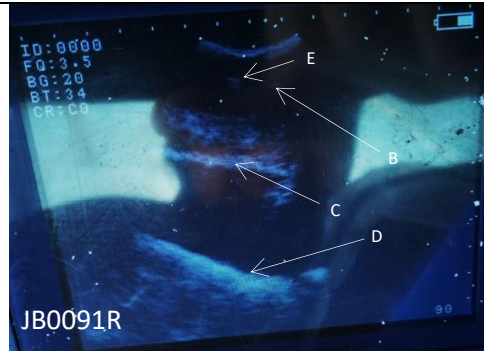
JB0090



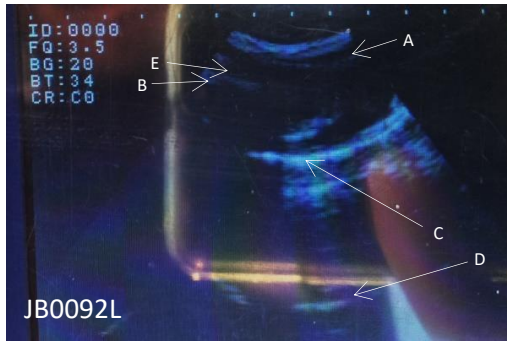
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JB0091



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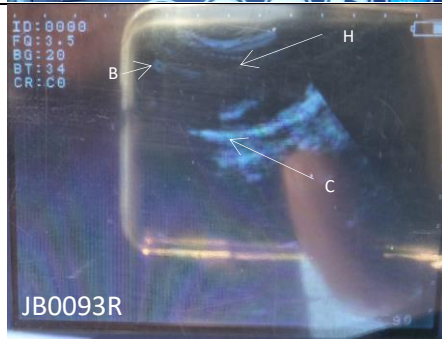


JB0092

JB0092L

JB0092R

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JB0093

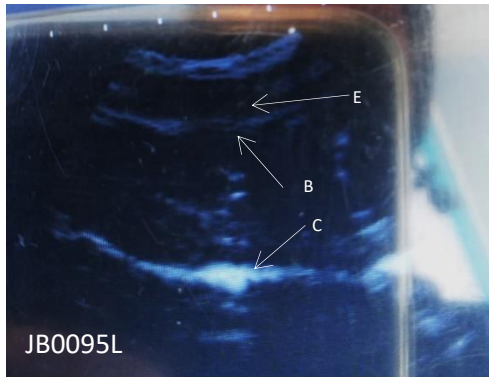
JB0093L

JB0093R

0

JB0094

M



JB0095

JB0095L

JB0095R

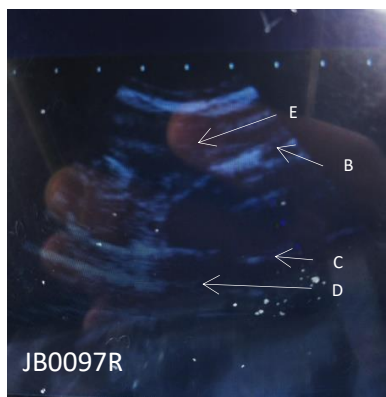
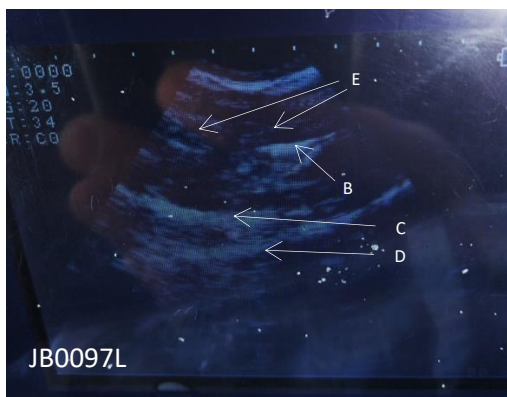
JB0096

M

M

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M

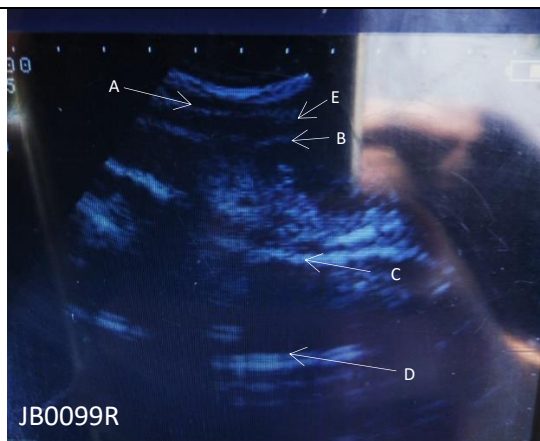


JB0097

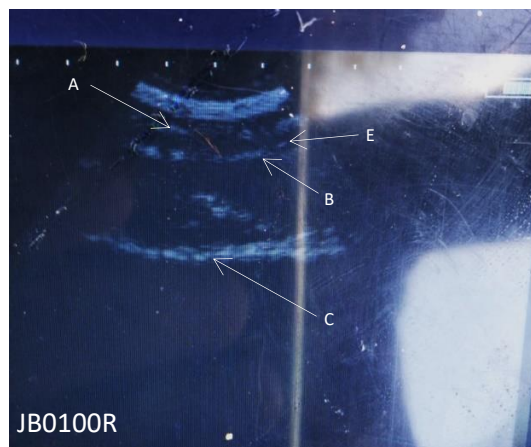
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JB0098

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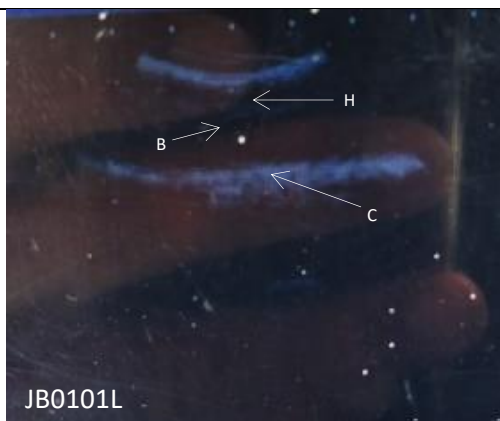
JB0099



JB0100

JB0100R

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JB0101

JB0101L

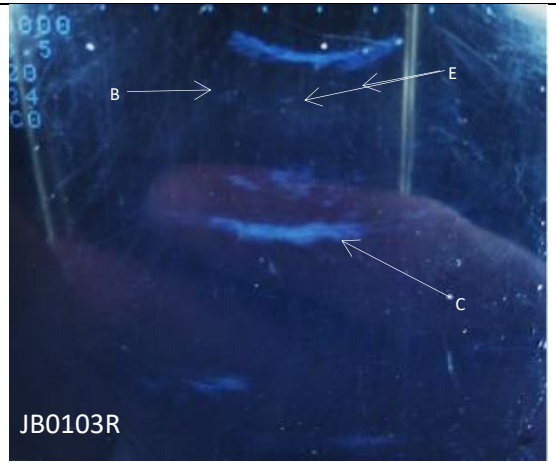
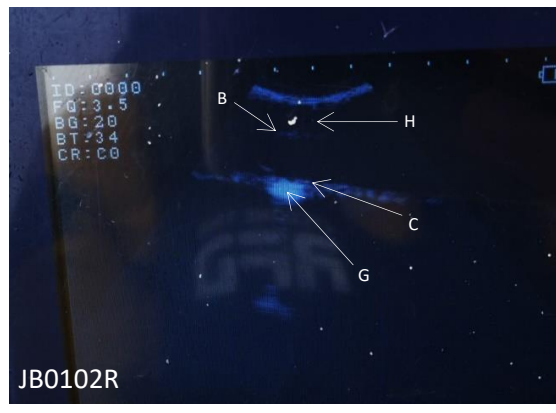
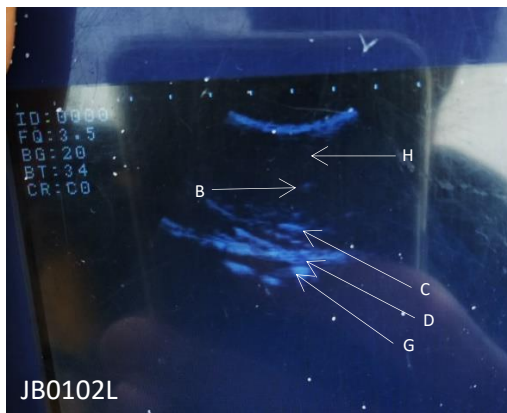


JB0101L

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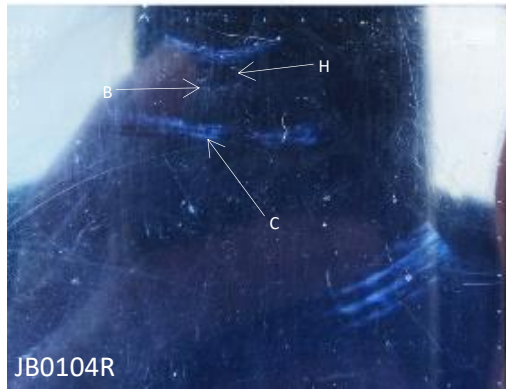
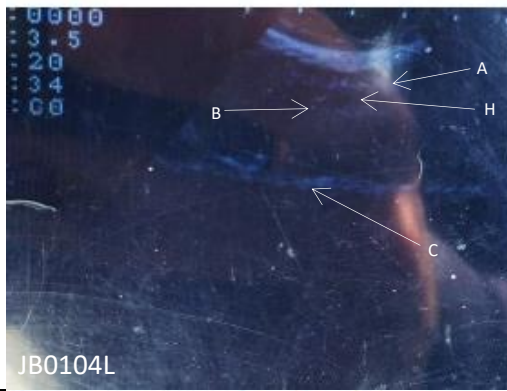
JB0102

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JB0103

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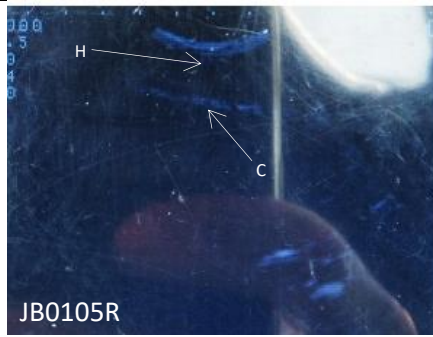
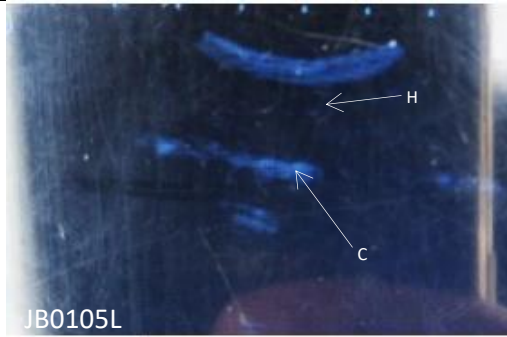


JB0104

JB0104L

JB0104R

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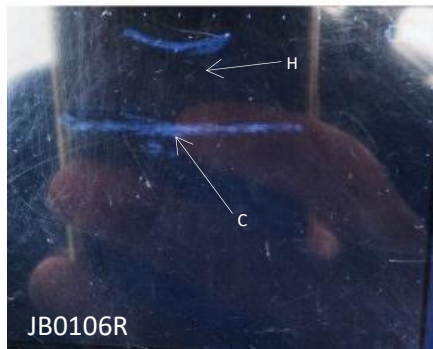


JB0105

JB0105L

JB0105R

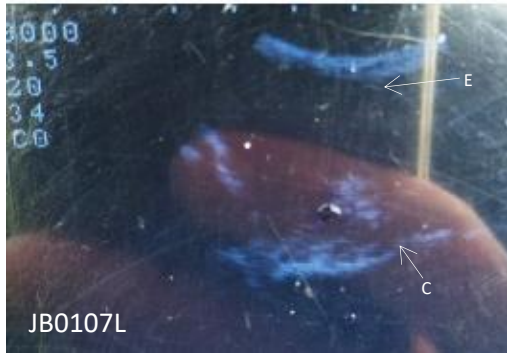
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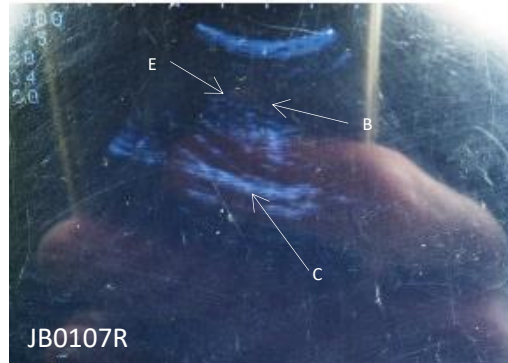
JB0106

JB0106R

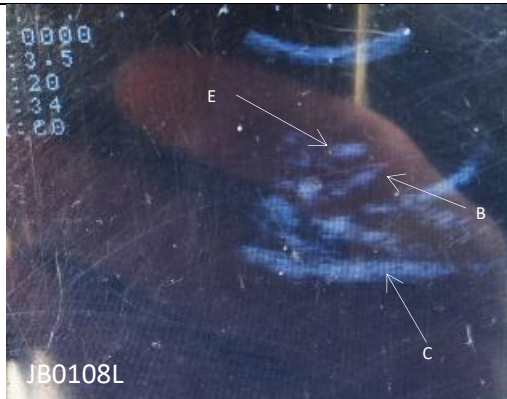
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JB0107



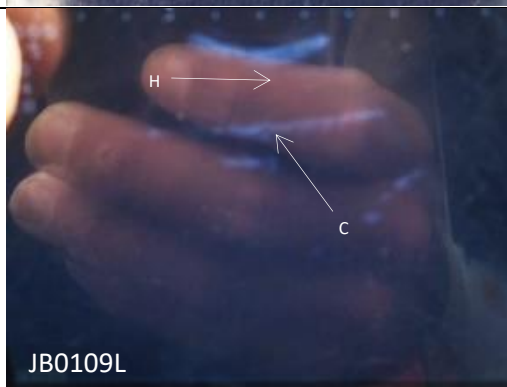
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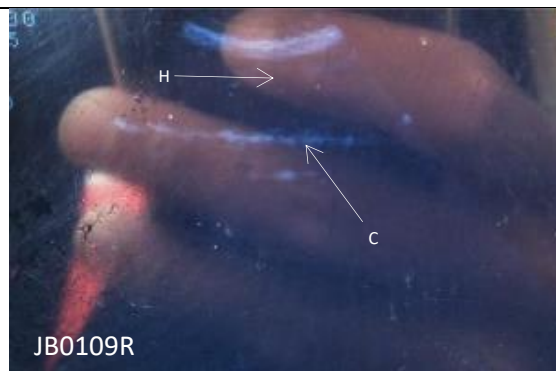
JB0108



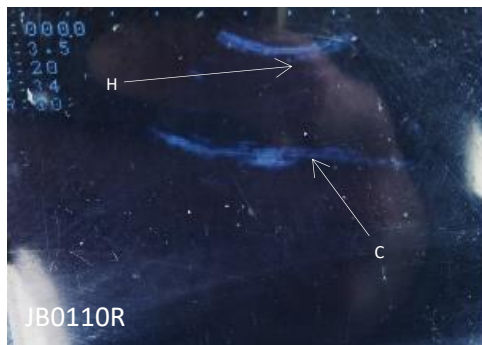
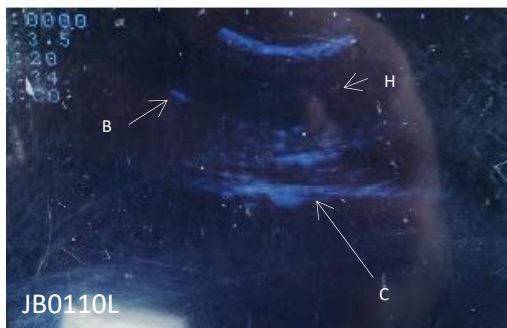
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JB0109

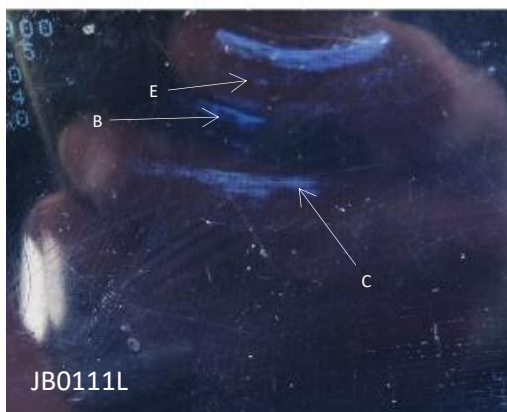


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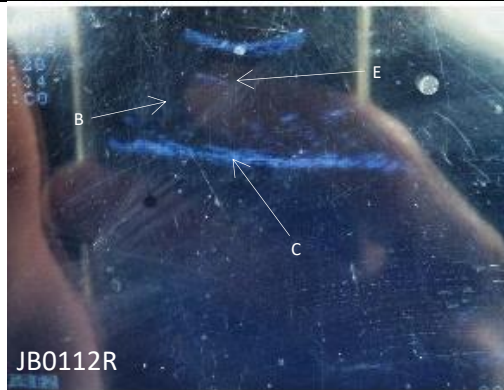
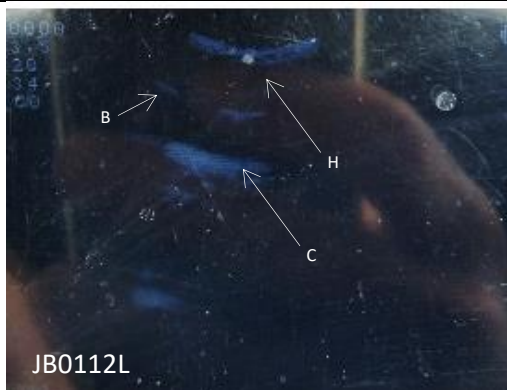
JB0110

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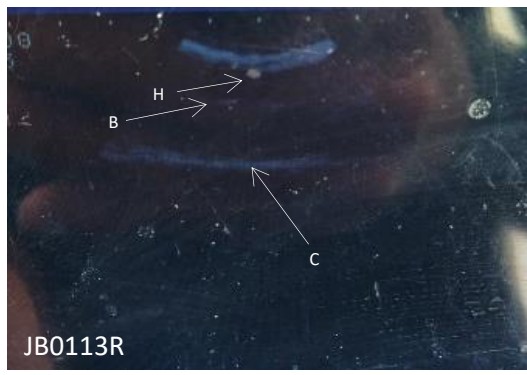
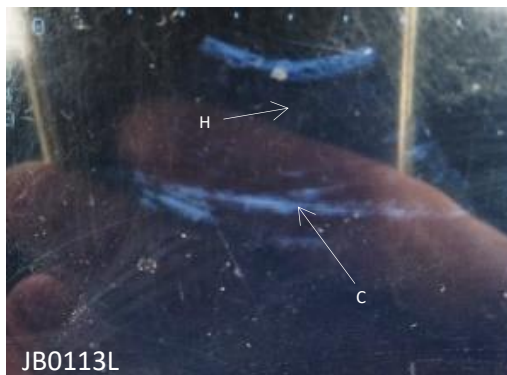
JB0111

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JB0112

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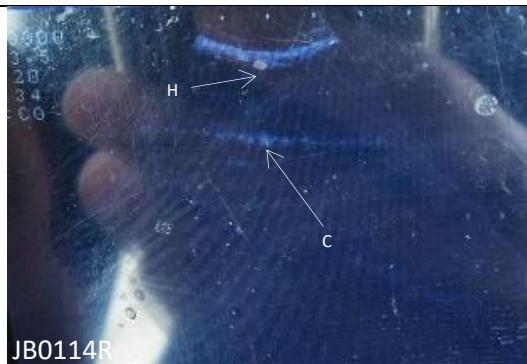
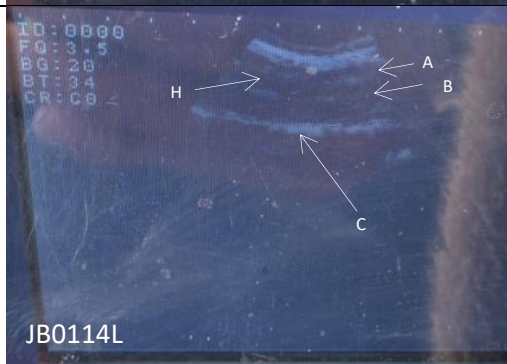


JB0113

JB0113L

JB0113R

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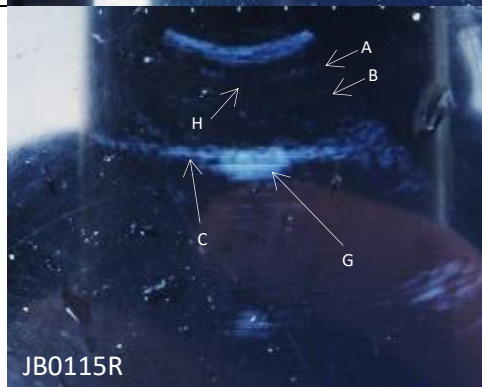
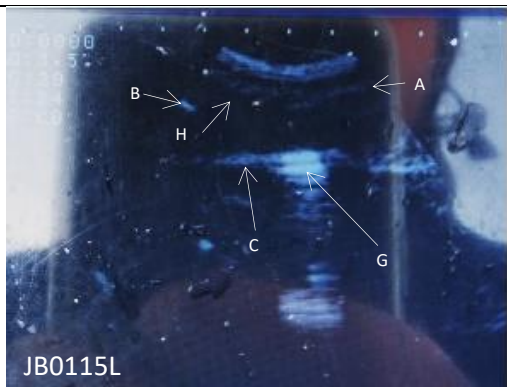


JB0114

JB0114L

JB0114R

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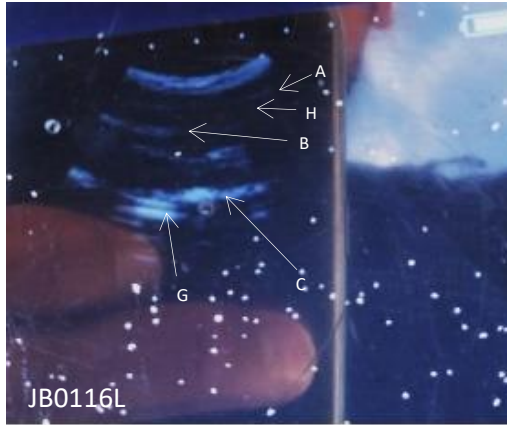


JB0115

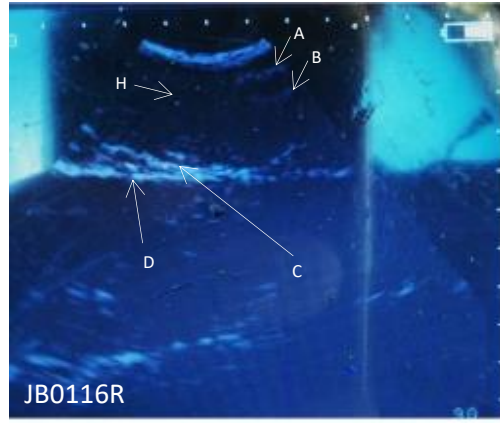
JB0115L

JB0115R

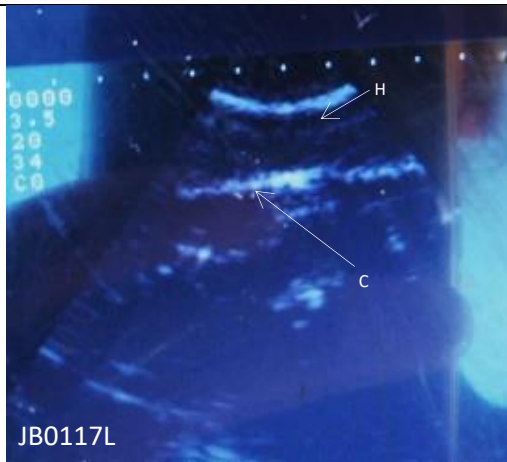
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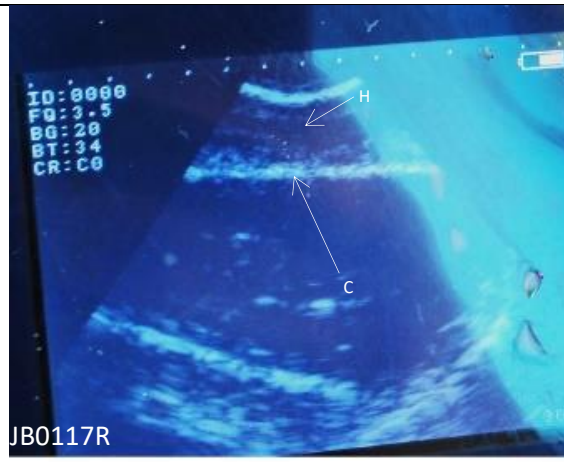
JB0116



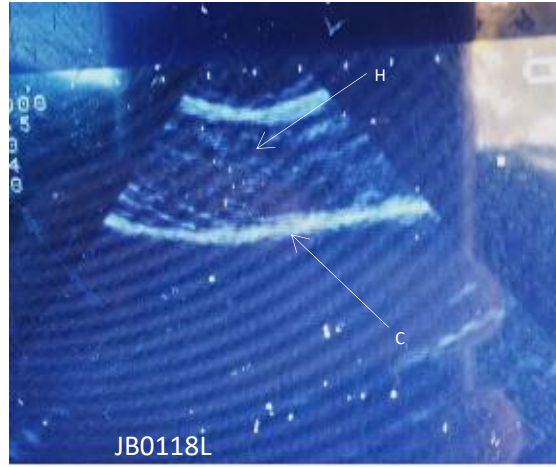
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JB0117



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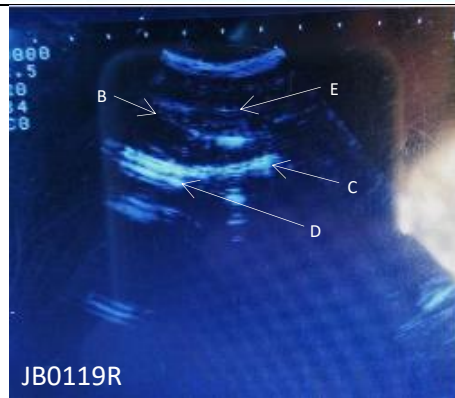
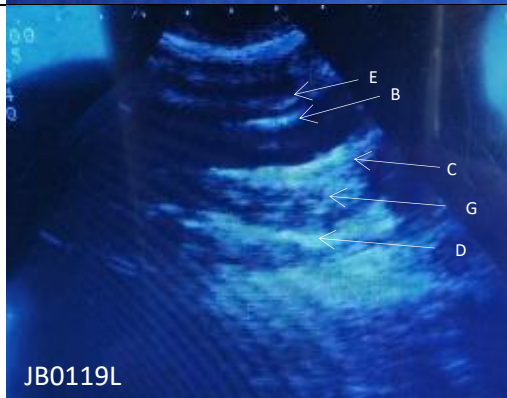


JB0118

JB0118L

JB0118L

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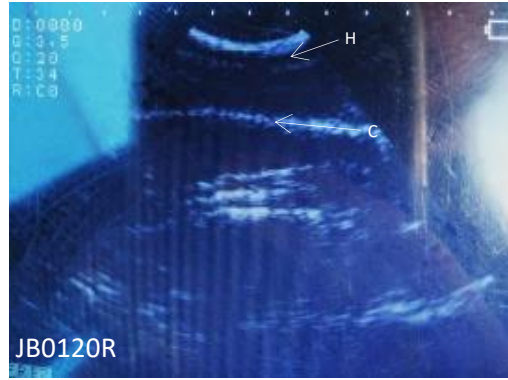
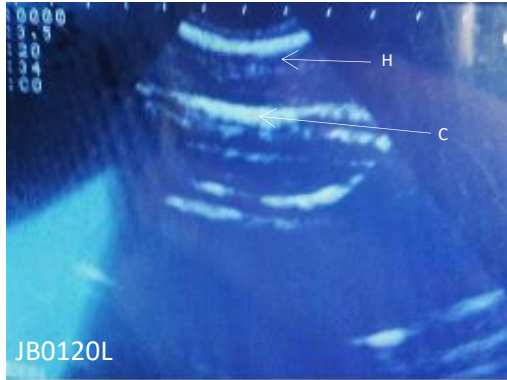


JB0119

JB0119L

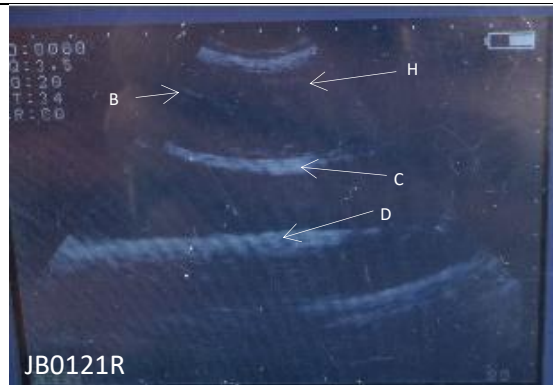
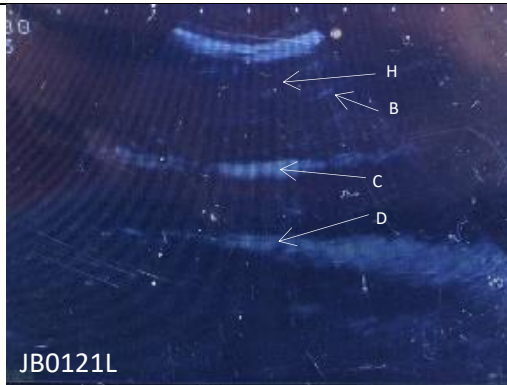
JB0119R

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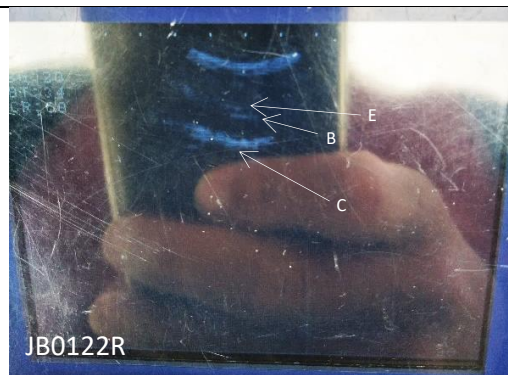
JB0120

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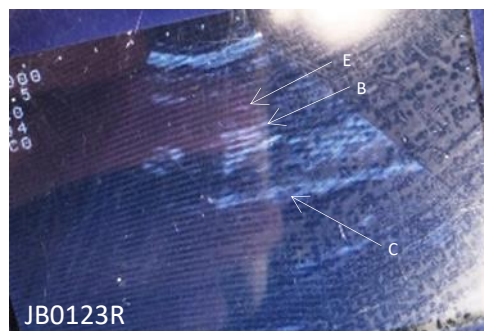
JB0121

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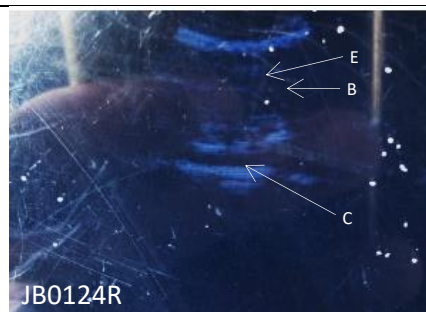


JB0122

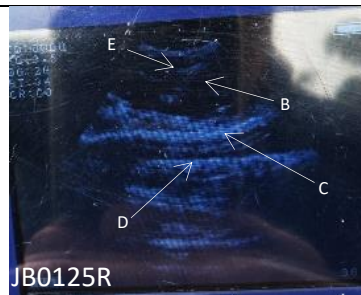
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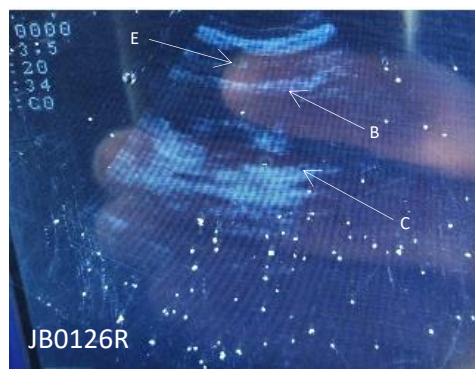
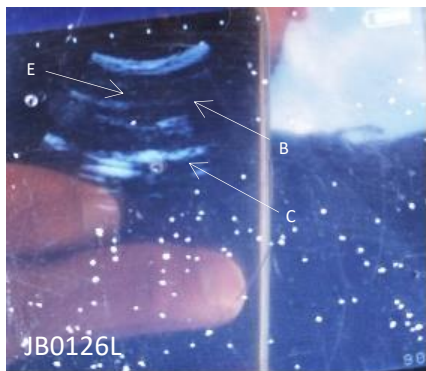
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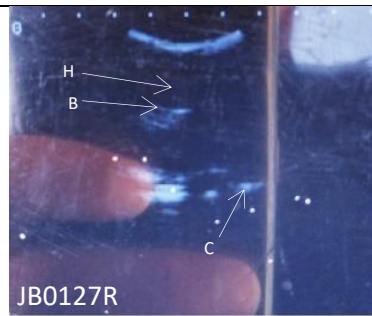
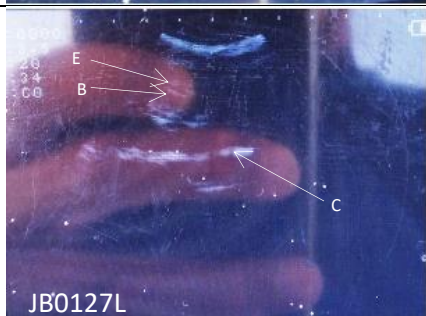


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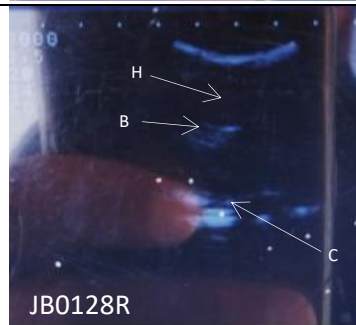
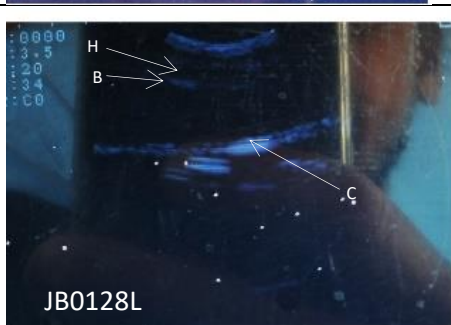
JB0126

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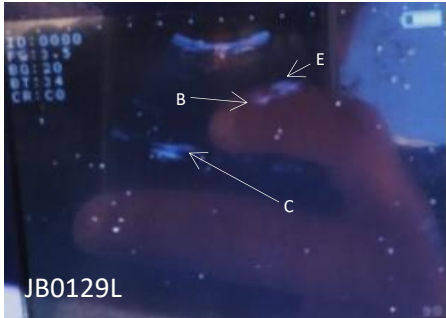
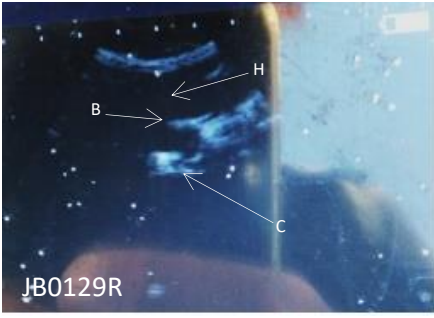

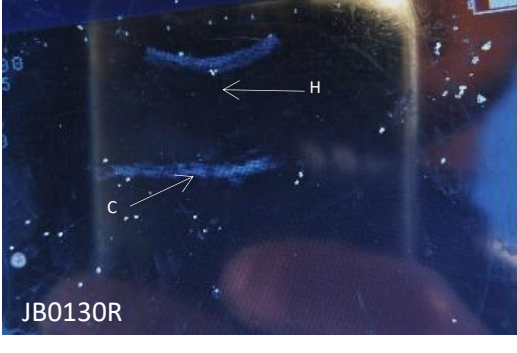
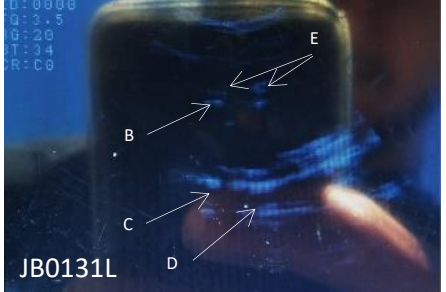
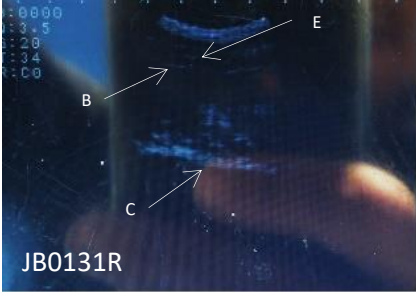
JB0127


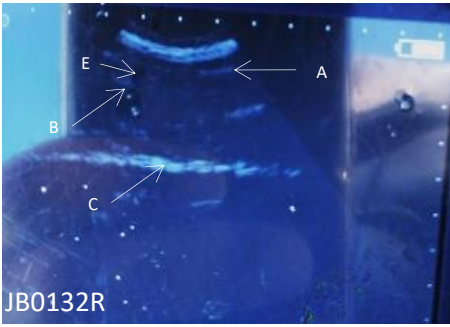

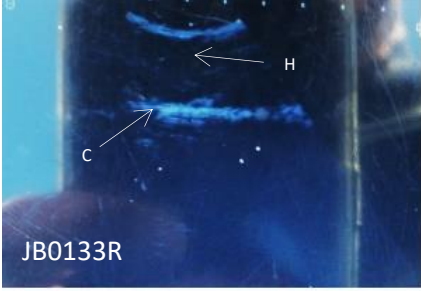

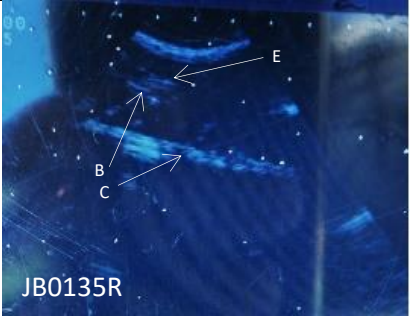
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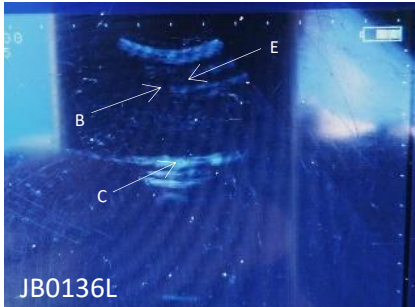
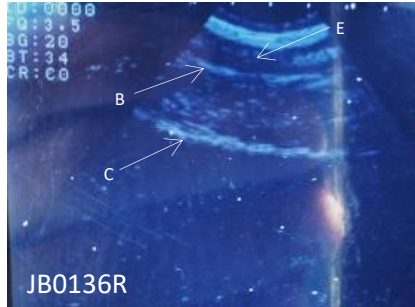
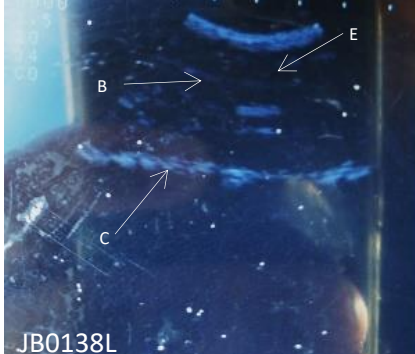
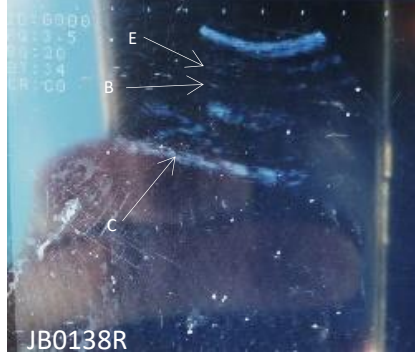
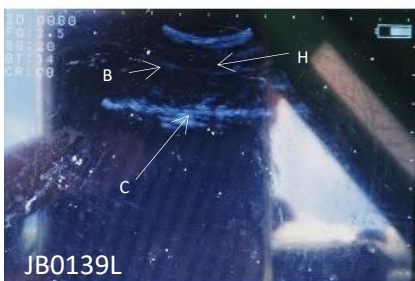
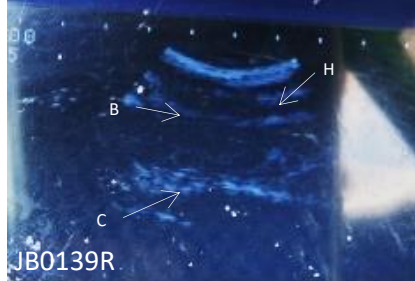


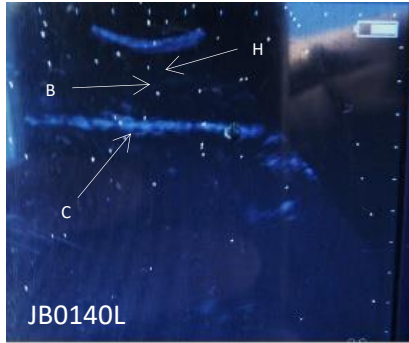
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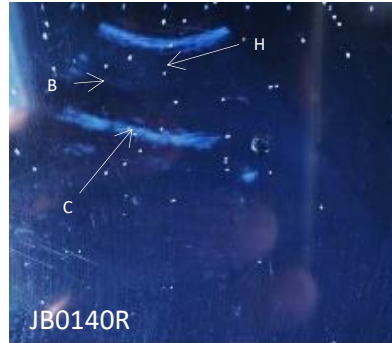
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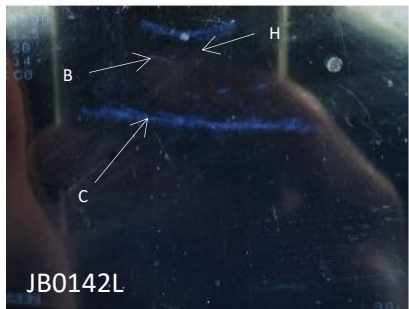


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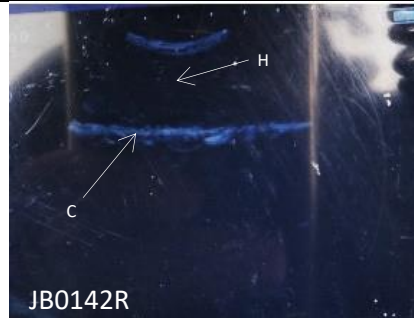
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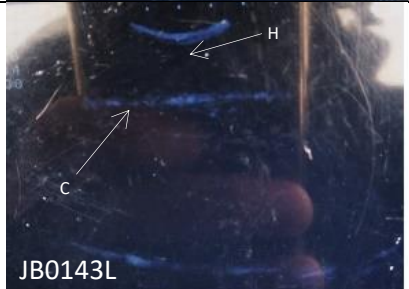
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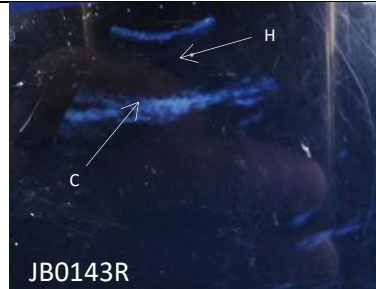
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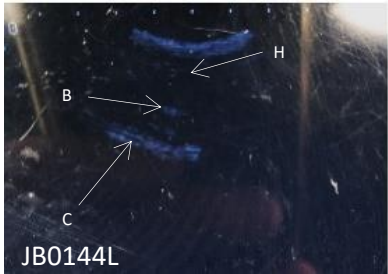
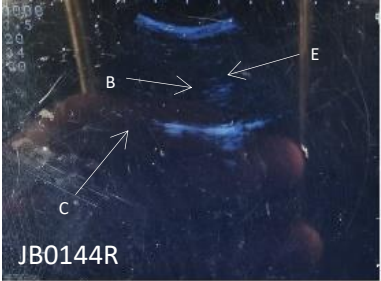
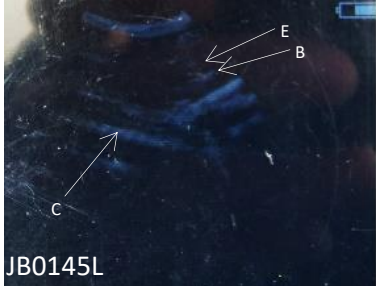

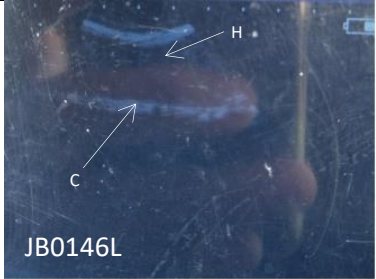
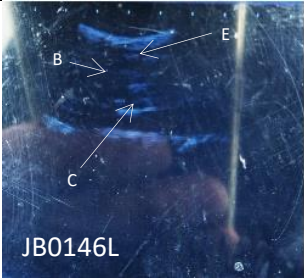


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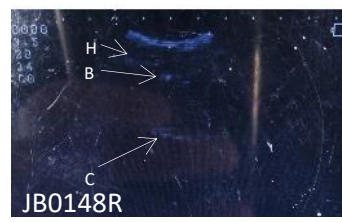
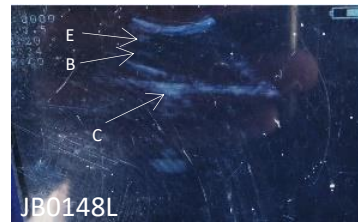


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JB0148



1

Appendix 5

Model output from chapter 6.3.1: the effect of environmental variables on the spotting rate of shark analogues

Generalized linear mixed model fit by maximum likelihood (Laplace Approximation) ['glmerMod']

Family: binomial (logit)

Formula: Saspotted ~ Depth * Sun * WindSpeed + (1 | Wave) + (1 | Lifeguard_ID)

Data: dat

Control: glmerControl(optimizer = "bobyqa", optCtrl = list(maxfun = 2e+05))

AIC	BIC	logLik	deviance	df.resid
-----	-----	--------	----------	----------

1152.2	1201.4	-566.1	1132.2	1010
--------	--------	--------	--------	------

Scaled residuals:

Min	1Q	Median	3Q	Max
-3.8091	-0.7387	0.3266	0.6712	4.2674

Random effects:

Groups	Name	Variance	Std.Dev.
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Lifeguard_ID	(Intercept)	0.3270	0.5718
--------------	-------------	--------	--------

Wave	(Intercept)	0.1686	0.4106
------	-------------	--------	--------

Number of obs: 1020, groups: Lifeguard_ID, 20; Wave, 4

Fixed effects:

	Estimate	Std. Error	z value	Pr(> z)
(Intercept)	-0.60732	0.30497	-1.991	0.046435 *
DepthShallow	1.24972	0.23997	5.208	1.91e-07 ***
SunSunny	1.95722	0.35984	5.439	5.35e-08 ***
WindSpeed1	-2.84560	0.76339	-3.728	0.000193 ***
DepthShallow:SunSunny	-0.09162	0.53677	-0.171	0.864467
DepthShallow:WindSpeed1	1.63235	0.81739	1.997	0.045821 *
SunSunny:WindSpeed1	0.90366	0.82841	1.091	0.275347
DepthShallow:SunSunny:WindSpeed1	-0.47154	0.97688	-0.483	0.629308

Signif. codes: 0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1

Correlation of Fixed Effects:

(Intr) DpthSh SnSunny WndSp1 DpS:SS DS:WS1 SS:WS1

DepthShallw -0.424

SunSunny -0.279 0.379

WindSpeed1 -0.127 0.148 0.071

DpthShll:SS 0.189 -0.444 -0.591 -0.067

DpthShl:WS1 0.123 -0.288 -0.106 -0.909 0.129

SnSunny:WnS1 0.100 -0.148 -0.400 -0.903 0.250 0.839

DptS:SS:WS1 -0.100 0.247 0.331 0.755 -0.550 -0.837 -0.831

Linear Hypotheses:

Estimate Std. Error z value Pr(>|z|)

Shallow.Cloudy.LowWind - Deep.Cloudy.LowWind == 0	1.24969	0.23997	5.208
Deep.Sunny.LowWind - Deep.Cloudy.LowWind == 0	1.95716	0.35984	5.439
Shallow.Sunny.LowWind - Deep.Cloudy.LowWind == 0	3.11548	0.44512	6.999
Deep.Cloudy.HighWind - Deep.Cloudy.LowWind == 0	-2.84583	0.76355	-3.727
Shallow.Cloudy.HighWind - Deep.Cloudy.LowWind == 0	0.03654	0.33939	0.108
Deep.Sunny.HighWind - Deep.Cloudy.LowWind == 0	0.01533	0.23924	0.064
Shallow.Sunny.HighWind - Deep.Cloudy.LowWind == 0	2.33415	0.25745	9.066
Deep.Sunny.0 - Shallow.Cloudy.LowWind == 0	0.70747	0.34862	2.029
Shallow.Sunny.0 - Shallow.Cloudy.LowWind == 0	1.86578	0.43528	4.286
Deep.Cloudy.HighWind - Shallow.Cloudy.LowWind == 0	-4.09552	0.76583	-5.348
Shallow.Cloudy.HighWind - Shallow.Cloudy.LowWind == 0	-1.21316	0.3415	-3.552
Deep.Sunny.HighWind - Shallow.Cloudy.LowWind == 0	-1.23437	0.23216	-5.317
Shallow.Sunny.HighWind - Shallow.Cloudy.LowWind == 0	1.08445	0.24424	4.44
Shallow.Sunny.0 - Deep.Sunny.LowWind == 0	1.15831	0.4891	2.409
Deep.Cloudy.HighWind - Deep.Sunny.LowWind == 0	-4.80299	0.8208	-5.852
Shallow.Cloudy.HighWind - Deep.Sunny.LowWind == 0	-1.92063	0.45017	-4.266
Deep.Sunny.HighWind - Deep.Sunny.LowWind == 0	-1.94184	0.35689	-5.441
Shallow.Sunny.HighWind - Deep.Sunny.LowWind == 0	0.37698	0.35863	1.051
Deep.Cloudy.HighWind - Shallow.Sunny.LowWind == 0	-5.96131	0.86227	-6.914
Shallow.Cloudy.HighWind - Shallow.Sunny.LowWind == 0	-3.07894	0.5209	-5.911
Deep.Sunny.HighWind - Shallow.Sunny.LowWind == 0	-3.10015	0.44277	-7.002
Shallow.Sunny.HighWind - Shallow.Sunny.LowWind == 0	-0.78133	0.44275	-1.765
Shallow.Cloudy.HighWind - Deep.Cloudy.HighWind == 0	2.88237	0.78285	3.682
Deep.Sunny.HighWind - Deep.Cloudy.HighWind == 0	2.86116	0.76002	3.765
Shallow.Sunny.HighWind - Deep.Cloudy.HighWind == 0	5.17998	0.77218	6.708
Deep.Sunny.HighWind - Shallow.Cloudy.HighWind == 0	-0.02121	0.33143	-0.064
Shallow.Sunny.HighWind - Shallow.Cloudy.HighWind == 0	2.29761	0.35388	6.493
Shallow.Sunny.HighWind - Deep.Sunny.HighWind == 0	2.31882	0.23619	9.818
Shallow.Sunny.1 - Shallow.Cloudy.1 == 0			
Shallow.Sunny.1 - Deep.Sunny.1 == 0			

Appendix 6

Chapter 2 of this thesis has an **Altmetric Attention Score** of **261** which puts it in the top 5% of all research outputs scored by Altmetric. For a detailed breakdown of this score and relevant media including articles in Forbes, the Guardian and Mongabay please see:

<https://www.altmetric.com/details/28090310/news>

Although **Chapter 5 and 6** of this thesis are yet to be published, the research has received considerable media attention as outlined below.

TELEVISION & SOCIAL MEDIA

Media Agency

Link

ABC News	https://www.facebook.com/abcillawarra/videos/1190214147691103/ [Accessed 19 June, 2019]
Australia Wide (ABC)	http://www.australiaplus.com/international/youtu-be-sharksurvivors-australiawide-abc-161024/7959484 [Accessed 19 July, 2017]
Australia Wide (ABC)	http://iview.abc.net.au/programs/australia-wide/NU1601H037S00 [Accessed 19 July, 2017]
Win News Illawarra	https://www.facebook.com/WINNewsIllawarra/videos/1471851406164931/ [Accessed 19 June, 2019]
Kiama Council	https://www.facebook.com/kiamacouncil/videos/714105045409313/ [Accessed 19 June, 2019]
Global Challenges	https://vimeo.com/189400671 [Accessed 19 June, 2019]
University of Wollongong	https://youtu.be/pU9WkCCtmVw [Accessed 19 June, 2019]
Save Our Seas Foundation	https://www.facebook.com/saveourseasngo/videos/10157174738091729/ [Accessed May 28, 2018]
9 News Illawarra	https://www.facebook.com/9NewsIllawarra/videos/2030831367183122/
Self Published	https://www.projectairship.net/
ASFB science communication competition	https://d3112vvksyz7k7.cloudfront.net/submission-entries/submission_videos/000/005/249/original/

[project AIRSHIP shark spotting made simple1493381706.mp4?1493382002](http://projectAIRSHIP.com/shark-spotting-made-simple1493381706.mp4?1493382002)

PRESS

Media agency

Link

The Huffington Post

http://www.huffingtonpost.com/atlas-obscura/australias-newest-beach-s_b_12470976.html [Accessed 19 June, 2019]

The Illawarra Mercury

<http://www.illawarramercury.com.au/story/4214634/blimp-on-shark-patrol/?cs=300> [Accessed 19 June, 2019]

ABC news

<http://www.abc.net.au/news/2016-10-21/shark-attacks-blimps-and-drones-may-be-better-than-nets/7952560?sf39666023=1> [Accessed 19 June, 2019]

The Kiama Independent

<http://www.kiamaindependent.com.au/story/4217249/lofty-ambition-for-blimp-on-shark-patrol/> [Accessed 19 June, 2019]

Blog post (goodthingimpretty)

<http://goodthingimpretty.com/australian-blimp-to-cruise-the-beaches-on-summer-shark-patrol/> [Accessed 19 June, 2019]

Mental Floss

<http://mentalfloss.com/article/87303/lifeguard-builds-blimp-scan-australian-beach-sharks> [Accessed 19 June, 2019]

The Bugle

<https://thebugleonline.com/2016/10/20/meet-kiamas-shark-blimp/> [Accessed 19 June, 2019]

UOW media release

<http://media.uow.edu.au/releases/UOW222484.html> [Accessed 19 June, 2019]

The Inertia

<http://www.theinertia.com/surf/shark-spotting-blimp-tested-by-lifeguards-in-australia/> [Accessed 19 June, 2019]

Wave Length Magazine

<http://www.wavelengthmag.com/shark-spotting-blimp-takes-to-the-skies/> [Accessed 19 June, 2019]

Atlas Obscura

<http://www.atlasobscura.com/articles/australias-newest-beach-safety-tool-is-a-sharkspotting-blimp> [Accessed 19 June, 2019]

Southern Highland News	http://www.southernhighlandnews.com.au/story/4214634/blimp-on-shark-patrol/?cs=55 [Accessed 19 June, 2019]
Kiama Independent.	https://www.kiamaindependent.com.au/story/5115937/kiamas-shark-spotting-blimp-takes-to-the-sky/ [Accessed 30 May, 2017]
Illawarra Mercury	https://www.illawarramercury.com.au/story/5407338/real-time-shark-alert-success-at-kiamas-surf-beach/#slide=14 [Accessed 28 May, 2018]
The Bugle	https://thebugleonline.com/2017/12/14/project-airship-returns-for-summer/ [Accessed 30 May, 2017]
Kiama Independent	https://www.kiamaindependent.com.au/story/5115937/kiamas-shark-spotting-blimp-takes-to-the-sky/ [Accessed 30 May, 2017]
UOW Research Magazine	https://www.uow.edu.au/content/groups/public/@web/@raid/documents/mm/uow241958.pdf [Accessed 19 June, 2019]

RADIO

The Science Show – separate stories podcast. ABC Radio National presented by Robyn Williams. March 31, 2018.	http://radio.abc.net.au/programitem/pgjGBPzloV?play=true [Accessed May 28, 2018]
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